THE EFFECT OF WINERY WASTEWATER IRRIGATION ON THE PROPERTIES OF SELECTED SOILS FROM THE SOUTH AFRICAN WINE REGION

By

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DECLARATION

By submitting this dissertation, I declare that the entirety of the work contained therein is my own, original work, that I am the owner of the copyright thereof (unless to the extent explicitly otherwise stated) and that I have not previously in its entirety or in part submitted it at any other university for obtaining any qualification.

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SUMMARY

Due to an increase in wine production as well as an intensification of environmental legislation in South Africa, the need for guidelines for sustainable management of winery wastewater has increased. To address this, the first part of the study focused on the seasonal dynamics of the volumes and quality of undiluted winery wastewater. The soil chemical dynamics were monitored in two different soils that were irrigated with undiluted winery wastewater for three years. Over-irrigation with undiluted winery wastewater in combination with winter rainfall caused large amounts of cations, particularly K^+ and Na^+, to leach beyond 90 cm soil depth. Consequently, the leached elements are bound to end up in natural water resources over time. Irrigation with undiluted winery wastewater did not have a pronounced effect on soil $pH_{(KCl)}$. This was probably due to the decomposition of organic matter and the fact that the applied salts as well as dissolved organic or mineral acids leached beyond 90 cm depth.

The practical application of irrigation with diluted winery wastewater was assessed in a pot experiment. Irrigations were applied under a rain shelter over four simulated irrigation seasons. Four soils varying in texture were irrigated with winery wastewater that was diluted to 3000 mg/L chemical oxygen demand (COD). The four soils were irrigated with municipal water as a control. The rate of K^+ increase in the soil containing 20% clay was higher than in soils containing 13% clay, or less. This suggested that heavy soils will aggravate the risk of high K^+ levels. The risk of Na^+ accumulation increased linearly with the clay content in the soil. Low Ca^{2+} and Mg^{2+} concentrations in the diluted wastewater had no effect on the soil, irrespective of clay content. Irrigation with diluted winery wastewater increased soil $pH_{(KCl)}$ substantially in all soils over four simulated seasons. The soil pH increase was attributed to the addition of organic and mineral salts *via* the diluted winery wastewater to the soil.

The effect of simulated rainfall on soils irrigated with winery wastewater was also assessed in a pot experiment. Six soils with different clay content were irrigated with winery wastewater diluted to 3000 mg/L over one simulated irrigation season. Thereafter, good quality river water simulating winter rainfall was added to the pots. The rainfall was simulated according to the long term averages of the regions were the soils originated.

Leaching of cations, particularly K^+ and Na^+ occurred only from four of the six soils when winter rainfall was simulated. In one of the sandy soils, the simulated rainfall was too low to allow leaching. In the case of other soil where there was no leaching, high clay content of 35% in combination with low rainfall prevented leaching. Where three soils received the same amount of rainfall, more cations leached from the sandy soils compared to the two heavier soils. These trends indicated that leaching of cations was a function of soil texture and rainfall.

OPSOMMING

As gevolg van die toename in wynproduksie, asook ‘n verskerping in omgewingswetgewing in Suid-Afrika, het die behoefte vir riglyne vir volhoubare bestuur van kelderafvalwater ‘n belangrike aspek van wynproduksie geword. Om dit aan te spreek, het die eerste deel van die studie op die seisoenale dinamika van die volumes en gehalte van onverdunde kelderafvalwater gefokus. Die grondchemiese dinamika in twee verskillende gronde wat met onverdunde kelderwater besproei is,by twee verskillende kommersiële kelders oor drie seisoene gemonitor. Oorbesproeiing met die onverdunde kelderafvalwater, in kombinasie met winterreënval, het veroorsaak dat groot hoeveelhede katione, veral K^+ en Na^+, dieper as 90 cm gronddiepte geloog het. Die nagevolg hiervan is dat die geloogde elemente oor tyd in natuurlike water hulpbronne sal beland. Besproeiing met onverdunde kelderafvalwater het nie ‘n noemenswaardige effek op grond $pH_{(KCl)}$ gehad nie. Dis was heel waarskynlik te wyte aan die feit dat die organiese materiaal ontbind het, en dat die toegediende katione as opgeloste organiese of mineraal soute verby 90 cm diepte geloog het.

Die praktiese toepasbaarheid van besproeiing met verdunde kelderafvalwater is in ‘n potproef ondersoek. Besproeiings is onder ‘n reënskuiling oor vier gesimuleerde seisoene toegedien. Vier gronde met verskillende teksture is besproei met kelderafvalwater wat tot 3000 mg/L chemiese suurstof aanvraag (Eng. = chemical oxygen demand, of kortweg COD). As ‘n kontrole is die vier gronde met munisipale water besproei. Die K^+ toename in die grond wat 20% klei bevat het, was hoër as in gronde wat 13% of minder klei bevat het. Dit het aangedui dat die risiko van K^+ aansameling hoër is in swaarder gronde. Die risiko van Na^+ toename het reglynig toegeneem met klei inhoud in die grond. Lae Ca^{2+} en Mg^{2+} konsentrasies in die verdunde afvalwater het geen effek in die gronde gehad nie, ongeag die klei-inhoud. Besproeiing met verdunde kelderafvalwater het die grond $pH_{(KCl)}$ in al die gronde oor die vier gesimuleerde seisoene betekenisvol laat toeneem. Die pH toename in die gronde kon aan die toediening van organiese en mineraal soute deur middel van die verdunde kelderwater toegeskryf word.

Die effek van gesimuleerde winterreënval op gronde wat eers met verdunde kelderafvalwater besproei is, is ook met behulp van ‘n potproef ondersoek. Ses gronde

met verskillende kleiinhoude is vir een gesimuleerde besproeingseisoen met kelderafvalwater wat tot 3000 mg/L COD verdun is, besproei. Daarna is gesimuleerde winterreënval in die vorm van hoe kwaliteit rivierwater op die gronde toegedien. Die reënval is volgens die langtermyn gemiddeldes van die streke waar die gronde voorgekom het, gesimuleer. Loging van katione, veral K^+ en Na^+ het slegs by vier van die ses gronde tydens die gesimuleerde winterreënval voorgekom. In die geval van een van die sanderige gronde, was die gesimuleerde reënval te min om loging te veroorsaak. In die geval van die ander grond waar geen loging voorgekom het nie, het die hoë-klei inhoud van 35%, in kombinasie met lae winterreënval, loging verhoed. Waar drie gronde dieselfde hoeveelheid reënval ontvang het, het meer katione uit die sanderige grond in vergelyking met die twee swaarder gronde geloog. Hierdie tendense het aangedui dat loging van katione ‘n funksie van grondtekstuur en reënval is.

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CHAPTER 1. GENERAL INTRODUCTION AND PROJECT AIMS

1.1. INTRODUCTION

In South Africa, the number of wineries increased from under 300 in 1997 to almost 600 in 2010. The increases in production of wine has put more pressure on the natural resources such as vegetation, water and soil. The rapid growth in wine production and the intensification of the land use in most countries that produce wine needs to be matched with reducing the environmental impact of operations because winery wastewater is the most important aspect in wine cellars that could result in detrimental effects on the natural environment (Chapman *et al.*, 1995; Gajdos, 1998). Traditionally, wastewaters or effluents from wineries and other industries have been disposed of in evaporation ponds and in some cases in natural water courses (Chapman, 1995). The deterioration of water quality due to the disposal of wastewater into water bodies has resulted in the treatment of wastewater through land application all over the world (Cook *et al*, 1994). The increased demand for high quality water together with water shortages in arid and semi-arid regions have increased the water challenges to water management (Oron *et al.,* 1999). This has led to the development of guidelines for the management of wastewater and solid waste at existing wineries (Van Schoor, 2005).

In the past, most wineries in Australia used to dispose of their wastewater by means of evaporation and direct discharge into water courses (Chapman *et al.*, 1995). In South Africa, more than 70% of wine cellars dispose of their wastewater by means of land application using irrigation as the primary treatment (Van Schoor, 2000). Wastewaters from both distilleries and wineries are generated mainly from washing of equipment. In distilleries, wastewater can, in addition, be generated through distillation processes (Hazell, 1997). Composition of winery wastewater fluctuates on a daily basis and it also depends on how various wastewater streams are mixed (Levay, 1995). Besides having high contents of suspended solids in the form of grape skins and pips in the case of winery wastewater, both winery and distillery effluents have a high biological and chemical oxygen demand which could range between 1000-40000 mg/L with low pH ranging between 3 and 5 (Mulidzi, 2001).

Because of the intensification of environmental legislation, wine growers are expected to find solutions for the treatment or the reuse of their winery wastewaters (Van Schoor, 2001a). According to some authors, application of winery and distillery wastewater may have positive effects on soil. Papini (2000) found that direct land application of stillage as irrigation water and as fertilizer has positive effects such as: increase in soil pH, increase in water and mineral salt retaining characteristics and soil restoration. Although land application of winery and distillery wastewaters seems convenient, this practice may also have negative impacts on the natural resources, including the soil (Bond, 1998). Potential negative impacts include excessive nitrate leaching to the groundwater as well as the effects of increasing soil sodicity on current and future land uses (Bond, 1998). Increased sodicity may result in negative effects on the infiltration rate and hydraulic conductivity of the soil (Cameron *et al*., 1997). Where there are deep sandy soils, leaching of phosphorus to groundwater may be a potential limitation to sustainability (Papini, 2000).

Nitrogen and P may cause eutrophication in aquatic ecosystems in which surface waters are nutrient enriched (Dufault *et al*., 2008). The organic component of winery wastewater is of no benefit to the soil to which it is applied, instead it poses a serious pollution hazard to the soil and adjacent water bodies as it was found that those soils that exhibited a low water holding capacity (high permeability soils) could not retain the organic matter at the rates irrigation was applied at various wineries (Mulidzi, 2001). According to Bond (1998), soil scientists should use their knowledge in developing suitability guidelines for wastewater disposal through land application. In South Africa, present guidelines for wastewater irrigation are very general and there is an urgent need to improve them through predicting soil processes after wastewater irrigation. Land resource assessment is very important in determining its suitability for wastewater irrigation and the selection of the most suitable land.

In South Africa, this is the first study of its kind that looked at impact of winery wastewater on different soils commonly found in the wine region in order to establish suitability for wastewater application. The South African Wine Industry has co-funded various projects for the past 10 years in order to develop technologies and/or information that will contribute to responsible management of wastewater and more particular the use of winery wastewater by means of crop irrigation. Currently, the South African Department of Water and Sanitation is investigating the specific General Authorization aimed at wineries in order to allow beneficial crop irrigation. This PhD study forms part of the multidisciplinary project on the impact of wastewater irrigation by wineries on soils, crop growth and product quality which was funded by the Water Research Commission, Winetech, THRIP and the Agricultural Research Council.

The formulated hypotheses for the PhD study is as follows: Different soil types with different soil texture will react differently to winery wastewater irrigation. Winery wastewater containing high concentrations of cations will increase soil potassium and sodium after irrigation.

1.2. PROJECT AIMS

The overall aim of the study was to determine the impact of winery wastewater on different soils and the suitability of selected soils throughout the Western Cape for winery wastewater irrigation.

Objectives of the study:

(i)To determine the annual soil chemistry dynamics due to winery wastewater irrigation on existing and new grazing paddocks in order to develop recommendations for the management of winery wastewater through land application.

(ii)To determine the effect of winery wastewater irrigation on the chemical properties of four different soils.

(iii) Determining the vulnerability of selected soils in the different rainfall areas to degradation and excessive leaching after wastewater application. Develop recommendations for winery wastewater irrigation suitability's on high as well as low rainfall areas.

1.3. STRUCTURE OF THE STUDY

A literature review on the impact of irrigation with winery wastewater is presented in Chapter 2. This chapter also includes the production, composition and characteristics of winery effluents. It also covers the soils of the South African wine region, effects of wastewater irrigation on soil properties, legal requirements and guidelines for sustainable wastewater irrigation. The chapter concludes by looking at management systems available for mitigating winery wastewater quality. Chapter 3 explains the effect of winery wastewater on a grazing paddock that has been irrigated over 15 years and one that has only been irrigated with winery wastewater for 3 years. Chapter 4 outlines the design of the pot experiment to study the effect of irrigation with diluted winery wastewater on four differently textured soils., It describes the experimental layout, soil selection, packaging of soils to pots, irrigation system applied as well as irrigation volumes and analytical methods used for soil and winery wastewater analysis. Chapter 5 explains the effects of irrigation with diluted winery wastewater on cations, pH and phosphorus. Chapter 6 discusses the effect of simulated winter rainfall on six soils irrigated with winery wastewater. The chapter covers soil selection, composition of wastewater, simulated rainfall water and amount of rainfall applied. The results and discussion focus on the changes to the composition of soils after simulated winter rainfall. Chapter 7 gives general conclusions and recommendations.

CHAPTER 2. LITERATURE REVIEW ON THE IMPACT OF IRRIGATION WITH WINERY WASTEWATER

2.1. INTRODUCTION

Many countries are, or will shortly be, experiencing water shortages due to a combination of climate change and increasing demand for clean water (Sang *et al*., 2007). This shift coincided with an increase in the demand for irrigation water. To meet the irrigation demand, the practice of supplementing available water with untreated, as well as treated, industrial wastewater was implemented (Wang *et al*., 2007). Water is the most valuable resource that, if not handled with care, may lead to enormous shortages of good quality water in arid and semi-arid regions in the coming decades due to global climate change (Faisal Anwar, 2011). In most African and Asian cities, the growth in population has outpaced the sanitation and wastewater infrastructure, making it difficult to manage urban wastewater (Qadir *et al.*, 2010). Good quality water resources in arid and semi-arid regions are becoming scarcer due to quality water being prioritized or allocated for urban water supply (Jalali *et al.*, 2008).

The increase in urban population in most developing countries due to residents looking for better opportunities results in diversion of larger amounts of fresh water to domestic, commercial and industrial sectors, resulting in greater volumes of wastewater (Qadir *et al.*, 2010). The shortage of quality water led to an increasing demand to irrigate with water contaminated with salts (poor quality) such as saline groundwater, drainage water and treated wastewater (Jalali *et al.*, 2008). Irrigation in arid and semi-arid regions throughout the world has been linked with an increase in the salts concentration in the soil (Walker & Lin, 2008). The high demand for water in the agricultural sector has resulted in an increase in the reuse of treated and untreated municipal and industrial wastewaters (Wang *et al.*, 2007). Previous studies have indicated that wastewater irrigation has the potential to change soil properties (Wang *et al*., 2007). The capacity of the soil to handle an excess amount of water load will depend upon factors such as effluent composition, irrigation method, soil types and irrigation frequencies (Mulidzi, 2001). The environmental impact of irrigation with partially or treated wastewater has not yet been widely investigated (Cook *et al*., 1994). Due to the intensification of environmental legislation, wine growers are expected to find solutions for the treatment or the re use of their winery wastewaters (Van Schoor, 2001a). The objective of this literature review is to discuss the production, composition, treatment and application of winery wastewater to land.

2.2. PRODUCTION AND COMPOSITION OF WINERY WASTEWATER

Wine is produced by crushing and fermentating grapes which is then followed by the straining of skins and seeds, storage, clarification and maturation of the young wines (Mulidzi, 2001). This process is given schematically in Figure. 2.1.

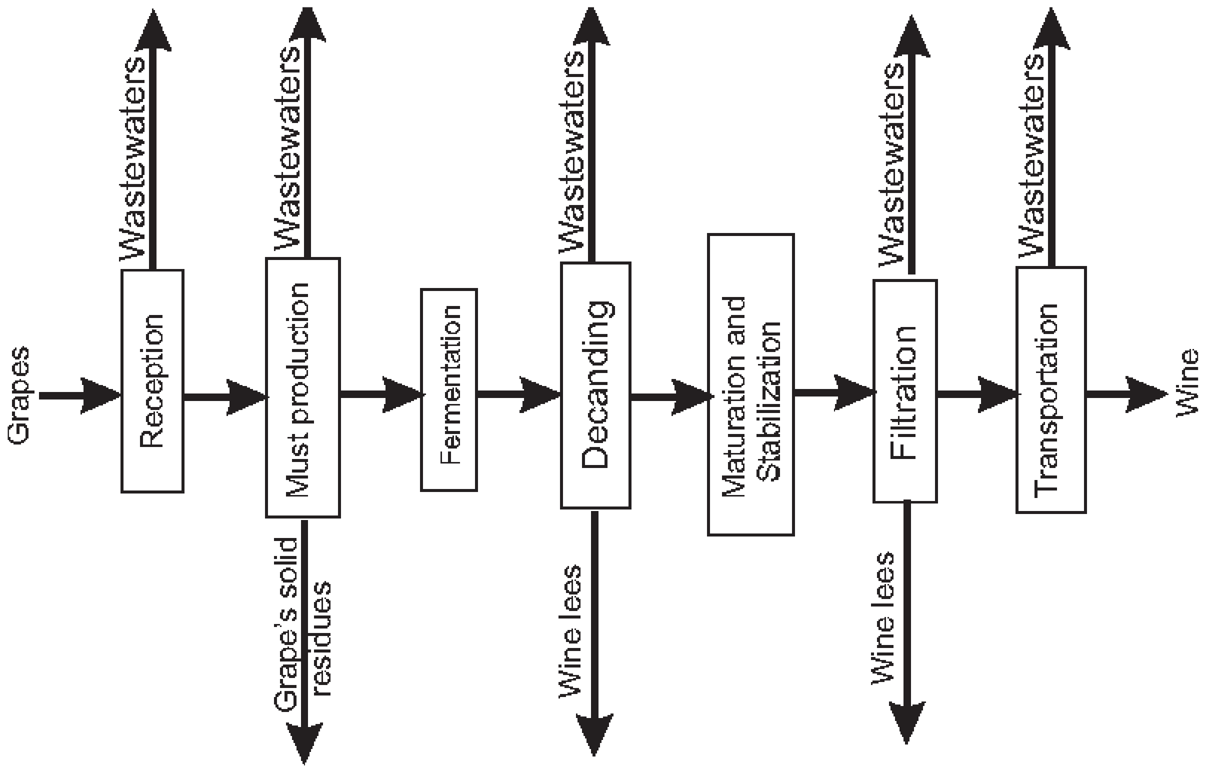

Figure 2.1. A schematic diagram of the general winemaking procedure followed in most wineries (Vlyssides *et al*., 2005).

According to Laurenson *et al*. (2010), wastewater generation is an inevitable aspect of wine production processes. Most wineries produce approximately 5 kiloliters of wastewater per ton of grape crushed (Chapman *et al*., 1995).

Winery wastewater is composed of mostly cleaning waste because wineries need to maintain cleanliness to avoid microbial contamination and spoilage (Mulidzi, 2001) In contrast, wastewater created by distilleries is generated through distillation processes (Hazell, 1997). Some water containing alkali salts is used in order to remove tartrates and other organic acids from inside the equipment as well as promoting earth filtering and ion exchange processes (Mulidzi, 2001). According to Chapman *et al.* (1995), winery wastewater comes from a number of sources that include: cleaning of tanks, ion exchange columns, hosing down of floors and equipment, barrel washing, spent wine and product losses, bottling facilities, filtration units, laboratory wastewater and storm water diverted into, or captured in the wastewater management system. Wine production is seasonal and can be divided into various stages (Table 2.1).

Composition of winery wastewater can even fluctuate on a daily basis, and also depends on how the various wastewater streams are mixed (Levay, 1995). Besides having a high content of suspended solids in the form of grape skins and pips in the case of wineries, effluents also have a high biological and chemical oxygen demand (COD) which can range from 1000 mg/L to 40000 mg/L. These values are in line with winery wastewater from Greece with COD of 3112 mg/L for white wines while for red wines it was in the range of 3997 mg/L (Vlyssides et.al., 2005). In addition, winery wastewater pH can be very low, and ranges between 3 and 5 (Mulidzi, 2001). However, most winery wastewater is characterized by high pH and high monovalent cation concentrations, in particular Na^+ and K^+. These two cations originate from potassium hydroxide (KOH) and sodium hydroxide (NaOH) detergents which are used for cleaning purposes (Lieffering & McLay, 1996).

Wine making processes results in generation of wastewaters of different quantities and qualities (Van Schoor, 2005). Winery wastewater and vinasse from Greece showed an acidic pH and high organic load. This combination makes the management of these wastes problematic hence different strategies need to be developed in order to reduce environmental risks (Bustamante et al., 2005). The effect of winery wastewater on legal quality parameters is summarized in Table 2.2.

Table 2.1 Description of winery wastewater generation and production periods at wineries (after Chapman *et al.*, 2001).

Period	Typical months of the year[1]	Description
Pre-vintage	January-February	Bottling, caustic washing of tanks, non-caustic washing of equipment in readiness for vintage
Early vintage	February-March	Wastewater production is rapidly rising to peak vintage flows and has reached 40% of the maximum weekly flow; vintage operations dominated by white wine production
Peak vintage	March-May	Wastewater generation is at its peak; vintage only operations are at a maximum
Late vintage	April-June	Wastewater production has decreased to 40% of the maximum weekly flow; vintage operations dominated by production of red wines; distillation of ethanol spirit may coincide with this period
Post vintage	May-September	Pre-fermentation operations have ceased; effect of caustic cleaning, ion exchange etc. is at its greatest, and wastewater quality may be poor.
Non-vintage	June-December	Wastewater generation is at its lowest-generally less than 30% of maximum weekly flows during vintage; wastewater quality is highly dependent on day by day activities

[1] In the southern hemisphere

Table 2.2. Major processes related to winery wastewater generation and their associated contribution to wastewater quality and quantity (after 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 neutralization	Up to 33%	Increase in Na^+, K^+, COD and pH	Increase in EC(1), SAR(1), COD(1) Variation in pH
Rinse water (tanks, floors, transfer lines, bottles, barrels, etc)	Up to 43%	Increase in Na^+, P, Cl^- and COD	Increase in EC, SAR, COD Variation in pH
Process water			
Filtration with filter aid	Up to 15%	Various contaminants	Increase COD and EC
Acidification and stabilization of wine	Up to 3%	NaCl	Increase COD and EC Decrease in pH
Cooling tower waste	Up to 6%	Various salts	Increase COD and EC
Other sources			
Laboratory practises	Up to 5-10%	Various salts, variation in pH, etc	Increase COD and EC

(1) EC= Electrical conductivity; SAR= Sodium adsorption ratio; COD= Chemical oxygen demand; NaCl=Sodium chloride

According to Levay (1995), winery wastewater contains the following:

- Simple organic acids, sugars and alcohols from grapes and wine. As a result, the wastewater has a high requirement for oxygen for biological decay.
- Moderate salinity, high concentrations of Na^+ relative to Ca^{2+} plus Mg^{2+} and low amounts of nitrogen and phosphorus relative to carbon.
- Inorganic components from the water supply, alkali wash waters and processing operations. Chemical fertilizers, pesticides and herbicides used in producing grapes are insignificant components of the effluent.
- Appreciable amounts of sulphur.

2.3. CHARACTERISTICS OF WINERY WASTEWATER

When a person wants to irrigate more than 10m^3 of winery wastewater, that person must register as a water user and up to 500m^3 may be irrigated per day if limits in Table 2.3 are met (Van Schoor, 2001a). When the COD of the wastewater is more than 400 mg/L but less than 5000 mg/L, 50 m^3 on any given day can be used for irrigation without a license. However, the water user may irrigate only above the 100-year flood line and no contamination to the surface or groundwater is allowed (Van Schoor, 2001a). It should be noted that in the legislation, there is no specified norm for the area that the water should be irrigated on.

Table 2.3. Limits for chemical oxygen demand (COD), faecal coliforms, pH, electrical conductivity (EC) and sodium adsorption ratio (SAR) for irrigation with wastewater in South Africa (after Myburgh & Howell, 2014).

Parameter	Maximum irrigation volume allowed (m^3/day)		
	< 50	< 500	< 2000
COD (mg/L)	5 000	400	75
Faecal coliforms (per 100 ml)	1 000 000	100 000	1 000
pH	6-9	6-9	5.5-9.5
EC (mS/m)	200	200	70-150
SAR	<5	<5	Other criteria apply

2.3.1. Biological oxygen demand (BOD)

A waste is normally characterized according to the effect its various contaminants have on the sewer or receiving waters (Gajdos, 1998). The BOD can be defined as a measure of organic waste that can be utilized by bacteria. While doing so, the bacteria utilizes oxygen thereby de-oxygenating the wastes, creating anaerobic conditions (Mulidzi, 2001). The disposal of untreated winery effluent can deplete soil oxygen thereby leading to anaerobic conditions in the soil (Levay, 1995). Prolonged anaerobic conditions have the potential to reduce the capability of soil microorganism's to decompose organic matter from winery wastewater leading to surface and groundwater pollution (Mulidzi, 2001).

2.3.2. Chemical oxygen demand (COD)

Chemical oxygen demand in winery wastewater can be defined as organic compounds that include organic acids, alcohols (ethanol) and phenolic compounds that has the ability to consume oxygen when they are degraded (Duncan et al., 1994). The phenolic compounds because of their ring structure, polymerize into long chained compounds that take long to degrade or treat (Shepherd & Grismer, 1997). Winery wastewater is mostly characterized by very high COD which is more than 15000 mg/L during vintage periods (Mulidzi, 2001). The high COD creates problems for discharging or disposal of wastewater (Shepherd & Grismer, 1997).

2.3.3. pH

Normally the pH of winery wastewater varies with relative concentrations of organic acids and caustic cleaning wastes which change quickly (Table 2.2). The pH of winery wastewater ranges between 4 and 8, but it is normally below 5.5 (Levay, 1995). The pH of winery wastewater depends on the activities within the cellar *i.e.* during vintage period when the crushing of grapes is taking place, the pH is mostly acidic but during non-vintage periods such as bottling, pH normally ranges between 5 and 8 (Levay, 1995). Most of the wineries use calcium hydroxide slurry to adjust the pH at the storage dam before irrigation (Hazell, 1997). The source of low pH in winery wastewater is the citric acid that is used to dissolve tartaric crystals (Van Schoor, 2000). The South African

General Authorization requires wastewater pH to be between 6 and 9 when irrigating (Table 2.3).

2.3.4. Sodium adsorption ratio (SAR)

The SAR indicates concentrations of Na^+ relative to Mg^{2+} and Ca^{2+} and the potential effects of Na^+ on soil structure (Levay, 1995). An excessive amount of Na^+ in winery wastewater relative to Ca^{2+} and Mg^{2+} has the ability to reduce the rate at which water moves into and through the soil as well as soil aeration (Ryder, 1994). The South African General Authorization stipulates that the SAR must not exceed 5 when irrigating with winery wastewater (Table 2.3).

2.4. SOILS OF THE SOUTH AFRICAN WINE REGION

The South African wine grape production region can be divided into four geographical zones *i.e.* Coastal, Breede River, Olifants River and Orange River regions (Bargmann, 2005). The three main soil types occurring predominantly in these vineyards are: residual and colluvial soils (soil movement had occurred due to weathering and chemical breakdown of local bedrock), alluvial soils (soils that have been deposited through action of water, mostly river waters) and aeolian soils (these sandy soils are deposited by wind action). Most soils in the Western Cape where vineyards are planted are low in P with the exception of soils along the Olifants River. Soils in the Coastal regions are dominated by the kaolin and sesquioxides clay minerals with increasing acidity while the inland soils are not acidic, and in most cases, contain free lime (Saayman, 2013).

2.5. EFFECTS OF WASTEWATER APPLICATION TO SOIL BY MEANS OF IRRIGATION OR PONDING ON SOIL PROPERTIES

Irrigation of winery wastewater has numerous effects on the soil's chemical and physical properties. This section of the literature review summarizes the major findings of previous studies.

2.5.1. Soil pH

Soil pH is one of the most important chemical properties of soils as it affects numerous chemical reactions, physical stability as well as plant nutrient availability. Application of winery wastewater has been shown to have a substantial effect on soil pH. A pH increase leads to an increased dissolution of organic matter which can induce dispersion in the soil (Faisal Anwar, 2011). High or low pH can lead to toxicity in macro- and micro-organisms as well as heavy metals solubility (Table 2.4). According to Lieffering and McLay (1996), soil pH and exchangeable Na^+ tend to increase when wastewater with high pH as well as high Na^+ concentrations has been used for irrigation. The accumulation of monovalent cations on the exchange sites has the ability to affect soil structure through clay dispersion and deflocculation processes. The presence of hydroxide solutions in wastewater has the ability to increase the soil cation exchange complex (CEC). High concentration of grey water contributes to higher EC due to the presence of salts in cleaning detergents (Faisal Anwar, 2011).

Table 2.4. Contaminants in winery wastewater, origins and likely environmental effects (after Kumar and Christen, 2009).

Contaminant class	Examples	Sources	Effects
Organics	Phenols, tannins, catechins, proteins, fructose, glucose, glycerol, ethanol, flavourings, citric acid, ethyl carbamate	Loss of juice, wine and lees, residues in cleaning waters and filters, solids reaching drains	Organism deaths, ecological function disruption, Odours generated by anaerobic decomposition, solubilisation of sorbed nutrients and heavy metals. Soil clogging
Nutrients	Nitrogen, Phosphorous, Potassium	Loss of juice, wine and lees, washings and ion exchange	Algal blooms, excess nitrate in water, high SAR
Salinity	Sodium chloride, Potassium chloride	Juice and wine, cleaning agents	Affects water taste, toxic to plants and animals
Sodicity	Sodium, potassium	Washing water	Degrades soil structure, toxicity to plants
Heavy metals	Al, Cd, Cr, Co, Cu, Ni, Pb, Zn, Hg	Al, Cu, piping and tanks, Pb soldering, brass fittings	Toxic to plants and animals
pH effects	Organic, sulphuric and phosphoric acids, sodium, magnesium and potassium hydroxides	Loss of juice, wine and lees, cleaning agents, wine stabilisation	Toxicity to macro and micro organisms, effect on solubility of heavy metals
Disinfectants	Sodium chloride, Sodium hypochlorite,	Sterilization of tanks, bottles, transfer lines	Formation of carcinogens
Soil cloggers	Microbial cells and grape residues, flocculating/coagulating agents, bentonite, diatomaceous earth	loss of lees and marc, floor cleaning, filtering, wastewater sludge	Reduction in porosity, light transmission, odour generation

2.5.2. Salinity

Salinity can be defined as soils with excess soluble salts in the soil solution thereby reducing growth of most crops. Wastewater irrigation leads to the addition of large amounts salts in the soil (Bond, 1998). In Australia, an annual application of 1000 mm of water with salinity of 500 mg/L of TDS adds 5 tons of salt per ha per year to the soil. To ensure sufficient leaching,of the accumulated salts, the soil should be permeable and this should be essential selection criteria for wastewater irrigation (Bond, 1998). Wastewater irrigation should be managed in a way that salts do not accumulate in the root zones and become toxic to plants (Table 2.4).

2.5.3. Sodicity

Soils characterized by high Na^{+} concentrations in their CEC usually have an exchangeable sodium percentage (ESP) of more than 15%. Winery wastewater contains high concentrations of Na^{+} relative to other cations and this could lead to degradation of soil structure (Table 2.4). High level of SAR in wastewater cause the ESP in the soil to increase (Bond, 1998).

High ESP causes to the deterioration of soil physical properties such as clay dispersion leading to soil structure breakdown, soil pore blockage and a decrease in permeability of the soil. Soil sodicity increases from wastewater irrigation may cause problems following cessation of irrigation and possibly change in land use (Bond, 1998).

2.5.4. Potassium

Potassium is one of essential nutrients found in soils for plant growth and it is a soil mineral highest in well drained or aerated soils. The land application of winery wastewater results in the accumulation of K^{+} in soil and leaching of Ca^{2+} and Mg^{2+} could lead to soil structure instability in the long term (Bond, 1998). The replacement of bivalent ions such as Ca^{2+} and Mg^{2+} by the monovalent ones like K^{+} during repeated irrigation can potentially lead to soil structural breakdown thereby affecting the soil hydraulic conductivity (Mosse *et al.*, 2011). Long term application of winery wastewater on pastures resulted in build-up of available K^{+} levels that has the potential to leach to the groundwater and other water sources (Christen *et al.*, 2010). Although the effects of high K^{+} ions applied to soil have not been researched extensively, it has been suggested that irrigation with potassium-rich wastewater could be advantageous to overall soil fertility but the long term application could result to alteration in the physico-chemical soil properties (Mosse *et al.*, 2011).

2.5.5. Clay dispersion and crusting

According to Sumner (1993), the likelihood of soil structural breakdown increases with soil pH increase, decreasing organic matter, increasing proportion of smectitic and illitic clays and increasing mechanical disturbance. Clay, organic matter and higher CEC in soil have effects on adsorption and protection of zinc and other elements but in soils with lower cation exchange capacities, the heavy metals are mostly adsorbed by plants (Bahmanyar, 2008).

2.5.6. Soil hydraulic conductivity

The movement of water through the soil is mostly measured as hydraulic conductivity (Hillel, 1980). Soil texture, pore continuity and proximity to water tables are some of the factors that determine the capacity of soil drainage which can assist to determine if the soil is suitable for land application of wastewater (Laurenson & Houlbrooke, 2012). According to Hillel (1980), sandy soil will have greater drainage capacity than the clay fine silt soils because, in general, finer soil texture has less pore continuity. To avoid loss of nutrients through deep drainage as well as surface runoff, the depth of winery wastewater irrigation during the time of application should be less than the soil water deficit (South Australian EPA, 2004).

2.5.7. Factors affecting infiltration rate and hydraulic conductivity

Winery wastewater contains many attributes that have the potential to reduce the hydraulic conductivity and infiltration rate of the soil on which it is irrigated (*Magesan et al*., 2000). Sodicity can cause the swelling and dispersion of clays resulting in changes to pore geometry thereby affecting hydraulic conductivity (Halliwell *et al*., 2001). Soil hydraulic conductivity and infiltration rate can be reduced through physical blocking of soil pores as a result of a high amount of suspended solids during continuous land application of wastewater (Magesan *et al*., 2000). According to Halliwell *et al*.(2001), repeated irrigation with wastewater containing suspended solids may result in the formation of restricting layer that can decrease the infiltration rate of the soil.

Halliwell *et al*. (2001) reported that wastewater irrigation causes reduction in soil hydraulic conductivity due to the following:

- Accumulation of suspended solids at the soil surface
- Blockage of the inter-soil spaces by suspended material such as colloidal clay and algal cell particles.
- Entrapped air bubbles.
- Formation of a biological mat or crust.
- Biological clogging including microbial extracellular polymeric materials such as polysaccharides.
- Collapse of soil structure due to organic matter dissolution.

2.6. DISPOSAL OF WINERY WASTEWATER THROUGH LAND APPLICATION

2.6.1. Background

More than 90% of wineries in South Africa dispose of their effluent by means of land application (Van Schoor, 2005). In order for the land treatment of winery wastewater to be sustainable, the treatment must have the ability to retain waste constituents in the soil and be effective in plant uptake of nutrients and contaminants that have been applied (Laurenson & Houlbrooke, 2012). If the conditions of the soil are suitable, the irrigation of crops with wastewater through land application could be practiced successfully if the salinity of the wastewater is low enough (Christen *et al*., 2010). The improvement of wastewater is needed in order to minimize health and environmental risks associated with wastewater irrigation. The effectiveness of wastewater reuse and disposal depends on the soil properties as well as the irrigation technology (Oron *et al*., 1999). Disposal of winery wastewater through land application has been practiced for many years as a treatment process, although it seems convenient, this practice may have a negative impact on the natural resources including the soil (Bond, 1998).

Papini (2000) found that direct land application of stillage as irrigation water and as fertilizer has positive effects such as an increase in soil pH, increase in water and mineral salt retaining characteristics, and restoration as well as maintenance of soil micro flora. The application of winery wastewater rich in soluble organic carbon to soils can, on the one hand lead to increased soil fertility through the conversion to soil organic matter while, on the other hand lead to the overloading of organic carbon which will be detrimental to soil health (Mosse *et al*., 2011).

2.6.2. Effects of land application

A survey by Mulidzi *et al*. (2009b) concluded that wineries differ regarding potential environmental hazard caused by their wastewater due to composition of effluents as well as the effectiveness regarding disposal practices and suitability of the disposal site. Wineries where Na^+ based cleaning agents such as sodium hydroxide are used, produce wastewater which can result in the accumulation of Na^+ in the receiving environment especially when the wastewater is discharged to the land (Mosse *et al*., 2011). Disposal of wastewater through land application on poorly drained soils will lead to salinisation and water logging thereby affecting the long-term sustainability of the site on which it is applied (Christen *et al*., 2010). Soil microorganisms also play an integral part of sustainable ecosystems and therefore any changes to microbial population as a result of winery wastewater irrigation need to be considered (Mosse *et al*., 2011).

In a recent study, the organic component of winery wastewater was of no benefit to the sandy soil to which it was applied (Mulidzi, 2001). Furthermore, soils that exhibited a low water holding capacity (high permeability soils) could not retain the organic matter at the rates that irrigation was applied at various wineries. Therefore, the application of winery wastewater to such soils poses a serious pollution hazard to the soil and adjacent water bodies. Although information on the impact of high K^+ concentration on soils is not available, there is the possibility that it could reduce the hydraulic conductivity of the soil on which it is applied (Mosse *et al*., 2011).

Improvements in the quality of water can be achieved in many developing countries, including South Africa, through primary treatment of wastewater especially where wastewater is used for irrigation (Mulidzi, 2001). Low cost treatment systems such as constructed wetlands and waste-stabilization ponds could also be helpful as secondary treatment systems (Qadir *et al.*, 2010). The integrating management of wastewater reuse in order to reduce or minimize treatment costs and improving agricultural productivity is gaining interest in many countries (Qadir *et al.*, 2010).

Within the framework of integrated natural resources, wastewater from different industries can be viewed as both effluent and a renewable resource. Most public authorities often do not have enough information and knowledge regarding the technical and management options available for minimizing and reducing environmental risks associated with wastewater irrigations (Qadir *et al.*, 2010).

2.6.3. Use of winery wastewater for crop irrigation

In South Africa, most wineries dispose of their effluent through irrigation of pastures in grazing paddocks. A study by Zingelwa and Wooldridge (2010) found that winery wastewater which contained Na^{+} concentrations of less than 400 mg/L did not have negative effect on the physiological status of vetiver and kikuyu grasses. In California, stored winery wastewater was used for vineyard irrigation during spring and summer (Ryder, 1994). More recently irrigation of Cabernet Sauvignon grapevines with winery wastewater diluted to 3000 mg/L and having SAR value of less than 10 did not have any effect on wine quality (Myburgh & Howell, 2014).

2.7. LEGAL REQUIREMENTS AND GUIDELINES FOR SUSTAINABLE WASTEWATER IRRIGATION

A survey on the composition of effluents from wineries in the Western and Northern Cape Provinces by Mulidzi *et al.* (2009a) found that none of the ten participating wineries complied with required environmental standards during the sampling period. According to Van Schoor (2001b), South African legislation requires that wine cellars adopt wastewater audit procedures as they do not have historic records of quality and volumes of their wastewater. Monitoring of wastewater impact on water resources, soil and vegetation should be compulsory (Van Schoor & Rossouw, 2004). A study investigating the sustainable use of greywater as an alternative water source for irrigation of gardens and small scale agriculture in South Africa found that, although there are no specific guidelines regarding greywater except the general wastewater which is governed by the National Water Act (NWA), of 1998, local authorities such as City of Cape Town have introduced policies which are still in draft stages (Rodda *et al*., 2011).

The South African irrigation guideline for agricultural water use require an SAR range of less than 2 to avoid sodium toxicity developing in plants sensitive to sodium whereas the EC should be less than 20 mS/m to ensure adequate infiltration rate. The recommended pH for irrigation water for agriculture should range from 6.5 to 8.4 (DWAF, 1996).

2.8. MANAGEMENT SYSTEMS FOR MITIGATING WINERY WASTEWATER QUALITY

According to Van Schoor (2001a), South African legislation does not permit the disposal of untreated winery wastewater into the natural water resources, which pressurized wine producers to manage their wastewater in a responsible manner. For the past ten years, the South African wine industry has invested money into research towards the sustainable treatment and management of winery wastewater (Mulidzi, 2005).

2.8.1. Constructed Wetlands

Constructed wetlands can be defined as natural wastewater treatment systems that have the potential to combine biological, chemical and physical treatment mechanisms for the improvement of water quality (Crites *et al.*, 1991). The rationale behind the use of constructed wetlands for treating winery wastewater is that wetlands are biological active ecosystems (Shepherd, 2002). According to Mulidzi (2005), the use of a constructed wetland to treat winery and distillery wastewater reduced the COD of the wastewater by *c.* 83% after only 14 days. A similar study resulted in 60% COD removal after seven days retention time (Mulidzi, 2006). Similar results were reported where constructed wetlands were used in California (Mulidzi, 2005). Constructed wetlands require low capital when compared to other treatment systems and they provide aesthetic value as well as habitat for wild life (Shepherd, 2002).

2.8.2. Up- flow anaerobic sludge blanket (UASB)

UASB involves the fermentation of organic matter into fatty acids that are volatile, alcohols, di-hydrogen and carbon dioxide by acidogenic bacteria

(Moletta, 2003). Wastewater that needs treatment is introduced in the bottom of the reactor and flows upward through a sludge blanket composed of biologically formed granules. The treatment process occurs when wastewater comes in contact with granules (Muller, 1998). The system is popular for biggest distilleries and wineries in France (Moletta, 2003). The micro-organisms are stored in the granules which are later suspended by the biogas produced as a result of by wastewater recirculation (Muller, 1998). The system had the ability to achieve COD removal of more than 90% and it can handle large quantities of wastewater but it needs trained personnel (Moletta, 2002).

2.8.3. The aerobic treatment system

For this type of treatment system, automatic machines supply air into the wastewater after every twenty or thirty minutes (Bloor *et al*., 1995). The technique application is limited to aerobic reactors only because of their capabilities to supply oxygen at a very low cost for the oxidation of organic matter to take place (Petruccioli *et al*., 2002). The system is used for large and medium size class wineries and is very effective in terms of COD removal (Eusebio *et al*., 2004).

CHAPTER 3. ANNUAL DYNAMICS OF WINERY WASTEWATER VOLUMES AND QUALITY AND THE IMPACTS OF DISPOSAL BY MEANS OF IRRIGATION ON SOILS

3.1. INTRODUCTION

Increasing wine production over the last two decades has necessitated wine producing countries to find sustainable winery wastewater management practices that address environmental concerns (Arienzo *et al.,* 2012). The use and availability of wastewater for irrigation has increased globally and the disposal of wastewater is governed by stringent legislations (Arienzo *et al.,* 2009a). Most wineries in South Africa dispose of their wastewater through land application (Van Schoor, 2001b). This is carried out by irrigating small areas of cultivated pasture with the wastewater or ponding, with the former being the more general practice (Mulidzi, 2001).

The use of winery wastewater for wine grape production is increasing, and it is therefore important to understand the environmental implication of such a practice (Laurenson *et al.,* 2012). Land application of winery wastewater results in the accumulation of K^+ and Na^+ in the soil and leaching of Ca^{2+} and Mg^{2+}, which could lead to the long term instability of soil structure (Mosse *et al*., 2011). The replacement of bivalent ions such as Ca^{2+} and Mg^{2+} by monovalent ones such as K^+ and Na^+ during continuous or long term repeated irrigation can potentially lead to the breakdown of the soil structure, thereby affecting the hydraulic conductivity of the soil (Lieffering & McLay, 1996). Long term application of winery wastewater on pastures resulted in the build-up of available K^+ levels that had the potential to leach into the groundwater and other water sources (Christen *et al*., 2010). Although the effects of having high K^+ concentrations in winery wastewater applied to the soil have not been researched extensively, it has been suggested that irrigating with K-rich wastewater could be advantageous to overall soil fertility, but long term application could result in the alteration of physicochemical soil properties (Mosse *et al*., 2011). A study by Arienzo *et al.* (2012) on effects of Na^+ and K^+ on soil hydraulic conductivity at a winery wastewater disposal site found that application of wastewater with high amounts of K^+ and Mg^{2+} resulted in loss of soil structural stability, as well as reduced hydraulic conductivity.

The current trend to replace sodium hydroxide with potassium based cleaning detergents in the cellars has the ability to increase levels of potassium in winery wastewater (Arienzo *et al.,* 2012). Accumulation of high levels of potassium in the soil is also regarded as a potential problem by regulators and the wine industry, because of the negative effect on soil structure and salt accumulation (Mulidzi *et al*, 2009b). According to Arienzo *et al.* (2009a), disposal of winery wastewater through land application has the potential to increase levels of soluble K^+ and the potassium exchange percentage (EPP) in soils since most K^+ in wastewater is immediately available. Soils with low clay content retained less K^+ in the exchangeable form, while soils with higher clay content retained K^+ to a much greater extent (Smiles & Smith, 2004). Another study showed that application of winery wastewater with K^+ and Na^+ concentration of about 400 mg/L on pastures and woodlots resulted in accumulation of available K^+ levels of 1400 mg/kg over a long term (Kumar & Kookana, 2006).

The actual amounts and the ratios between the four dominant basic cations, namely Ca^{2+}, Mg^{2+}, K^+ and Na^+, adsorbed on the soil exchange complex are important with regard to soil chemical and physical conditions, as well as plant nutrition. Adequate potassium is, for example, important for grapevine performance and K^+ deficiencies will cause low yields (Raath, 2012). On the other hand, excessive K^+ levels cause poor wine quality in terms of low acidity and poor colouring of red wines (Kodur, 2011). The K^+ exchange reactions with Ca^{2+}, Mg^{2+} or Na^+ on clay minerals and soils have been extensively studied. Some studies have shown soil preference for K^+ over Ca^{2+}, Mg^{2+} and Na^+ (Arienzo *et al.,* 2009a). Although limited research data exist on the effects of K^+ on structure stability, it seems that high levels of exchangeable K^+, similar to Na^+, can reduce soil hydraulic conductivity and water infiltration rate (Quirck & Schofield, 1955). The exchangeable cation composition in the soil is extremely important due to the different impacts of different cations with regard to dispersion and flocculation of soil colloids. Dispersion leads to degradation of soil structure, which causes problems such as soil crusting (surface sealing) and slaking that can lead to low water infiltration rates, low hydraulic conductivity, poor aeration, poor root development and functioning (Laker, 2004). High levels of Na^+ in the soil causes soil dispersion. Dispersion actually occurs when high-

Na soils are irrigated with fresh relatively low salinity water. It was previously believed that problems occur only at exchangeable sodium percentage (ESP) above 15. Research in Australia and South Africa has shown that in some soils Na^+ causes problems at much lower ESP values, even as low as 5, with the critical value varying between soils (Arienzo *et al.,* 2009b; Bond, 1998; Laker, 2004).

Application of winery wastewater that contains high concentrations of bicarbonate cleaning products has the potential to increase soil pH when applied to land (Laurenson & Houlbrooke, 2012). Soil pH increase due to crop residues application is attributed to addition of cations such as K^+, Na^+, Mg^{2+} and Mg^{2+} with plant materials (Yan *et al*., 1996). Disposal of winery wastewater containing high levels P can increase the concentration of dissolved P in runoff. This risk is greatest when rainfall or irrigation occurs immediately after application (Mulidzi *et al*., 2009b). Only few, plant species require more than 50 mg/kg soil P (Bingham, 1966). High levels of soil P due to winery wastewater application could create problems such as poor nodulation in legumes, zinc and copper deficiencies, as well as interference with sugar metabolism (Mulidzi *et al*., 2009a). When high levels of plant available P from the wastewater reach the fresh water streams, phosphate and organic phosphates are released. The latter can be assimilated by algae, plants and bacteria. Such water poses health hazard to humans (Corell, 1998).

In terms of the Department of Water Affairs General Authorisations (2013), most of South African wineries would not qualify to discharge their untreated wastewater into natural water resources. Where the disposal of winery wastewater is through land application, the following requirements as stipulated in the General Authorization must be met.

Up to 500 m^3 of wastewater may be irrigated for crop production, including grazing, on any given day provided that:

- The electrical conductivity (EC) is less than 200 (mS/m).
- The pH is between 6 and 9.
- The sodium adsorption ratio (SAR) does not exceed 5.

- The chemical oxygen demand (COD) is less than 400 mg/L. If the COD is higher than 400 mg/L, but less than 5 000 mg/L, irrigation (after registration) may not exceed 50 m^3 on any given day.

The composition of winery wastewater changes throughout the year. The large variability in volume and concentration of winery wastewater is associated with different practices that occur during different times of the year. Winery wastewater quality is usually at its worst when vintage operations are dominated by the production of red wines (Conradie *et al*., 2014). High pollution loads from July until November are associated with bottling of white wines, putting red wines to barrel and the filtering of previous year's red wines. In the Southern hemisphere, harvest is from end of January until beginning of April. Winery wastewater produced during harvest will typically contain higher levels of COD and salts than wastewater produced outside the harvest period (Kumar & Christen, 2009). Concentrations of COD and salts in winery wastewater fluctuate according to winery operations, and reaches a maximum when grapes are crushed (Laurenson *et al*., 2010). The lowest COD values usually occur in December and January (pre-harvest) and June and July (mid-winter) (Mulidzi *et al.,* 2009b). Peak periods of wastewater generation, as well as maximum levels of COD tend to coincide with peak harvest periods. Variation in the period of high COD reflected local differences in harvest period (Mulidzi *et al.,* 2009a). This variation also depends on the production period, as well as the unique style of winemaking of different wines. It must be noted that winery wastewater concentrations actually vary on a daily basis within a winery depending on the activities occurring at the time. The variation in concentrations is also determined by the amount of clean water used for specific processes, *e.g.* cleaning the floors and tanks. This work is unique as it focuses on real amount of winery wastewater applied per week and its direct environmental impact to a specific site.

The objective of this chapter is to investigate the annual dynamics of winery wastewater volumes and quality as well as the effect of winery wastewater irrigation on the chemical soil properties and potential environmental impacts at: (i) an existing grazing paddock at a winery near Rawsonville where wastewater has been applied for many years and (ii) a new paddock at a winery near Stellenbosch where no wastewater had previously been applied.

3.2. MATERIALS AND METHODS

3.2.1. Sites and soils

The experiment was carried out at two different sites, namely (i) at a winery near Rawsonville in an existing cultivated pasture grazing paddock where winery wastewater had been applied for over 15 years (-33.4137.7° 19.1920.3°) and (ii) at a winery near Stellenbosch in a newly cultivated pasture grazing paddock where no winery wastewater had been applied before (-33.4958.6° 18.4759.9°). Both sites are in the centre of wide flat plains. The grazing paddock at the Rawsonville winery had been irrigated with wastewater for more than 15 years. This site was considered to be representative of winery wastewater disposal through land application as practiced by most wineries in South Africa. The winery near Rawsonville annually crushes *ca.* 22 000 tons of grapes, whereas the one near Stellenbosch crushes 16 000 tons.

3.2.2. Characteristics of the soil at the Rawsonville site

The soils around Rawsonville were formed from the alluvium of the Breede Riverand are relatively young. The soil at the site selected for the study showed no clear stratification and contained a mottled subsoil, thus qualifying it for inclusion in the Longlands soil form (orthic A - E horizon - soft plinthic B horizon) or a Gleyic, Albic, Arenosol (IUSS Working Group WRB, 2014) (Fig. 3.1). The apedal soil consisted of fine sand. The B horizon showed few fine mottles with distinct contrast and brown colour (Appendix 3.1).

Figure 3.1. The Longlands soil form near Rawsonville showing no clear stratification.

Table 3.1. Particle size distribution in the 0-30 cm layer of the Longlands soil form in an existing grazing paddock at the Rawsonville winery.

Particle size (%)				
Clay (<0.002 mm)	**Silt (0.002-0.02 mm)**	**Fine sand (0.02-0.2 mm)**	**Medium sand (0.2-0.5 mm)**	**Coarse sand (0.5-2 mm)**
3.3	1	60	29	8

3.2.3. Characteristics of the soil at the Stellenbosch site

The soil at the winery near Stellenbosch was classified as a Kroonstad soil form, which consists of an orthic A- E-G horizon sequence (Soil Classification Working Group, 1991). According to the World Reference Base this soil would classify as a Gleyic, Albic, Planosol (IUSS Working Group WRB, 2014). Beneath the topsoil was a bleached, light grey, structureless, apedal sandy horizon (E/Albic horizon) to a depth of 50 cm (Appendix 3.2). This is an example of an E horizon that is yellow when moist (Fig. 3.2). Below this horizon is a gleyed clay layer (G horizon), indicating a zone of prolonged wetness due to poor drainage. Kroonstad soils commonly occur in the Stellenbosch winelands region.

Figure 3.2. The Kroonstad soil form at a winery near Stellenbosch showing its duplex character and waterlogged subsoil.

Table 3.2 Particle size distribution in the 0-30 cm layer of the Kroonstad soil form in an existing grazing paddock at the Stellenbosch winery.

Particle size (%)				
Clay (<0.002 mm)	**Silt (0.002-0.02 mm)**	**Fine sand (0.02-0.2 mm)**	**Medium sand (0.2-0.5 mm)**	**Coarse sand (0.5-2 mm)**
7	6	39	26	22

3.2.4. Experiment layout

At both sites, three 2 m x 3 m replication plots were demarcated for the experiment. Rain gauges were installed at each plot to measure the amount of wastewater applied. A two litre plastic bottle was attached to each rain gauge in the irrigation site in order to collect the overflow wastewater when the rain gauge was full (Fig. 3.3). Three rain gauges were also installed outside the wastewater demarcated area for measuring the rainfall.

Figure 3.3. Rain gauge with attachment to catch overflow for measuring the volume of wastewater applied to a replication plot at a winery near Stellenbosch.

3.2.5. Application of winery wastewater to the soils

At both sites, an overhead sprinkler was connected to the main wastewater line where the winery disposes its wastewater through land application. The wastewater received only preliminary treatment, *i.e.* screening to remove coarse particles, addition of lime to increase the water pH followed by settling of solids in a pond. The water treatment was carried out by the wineries. No irrigation scheduling was implemented. At both wineries all wastewater was disposed of through sprinkler irrigation. The amount of wastewater applied as well as rainwater was recorded on a weekly basis. At both sites, field measurements commenced on 1 March 2011 and were terminated on 30 November 2013.

3.2.6. Wastewater sampling and analysis

Winery wastewater sampling started in April 2011 at both wineries. Winery wastewater samples were collected from the rain meters once a week and analysed for chemical composition. The COD of the water was measured by the Soil and Water Science Division at ARC Infruitec-Nietvoorbij near Stellenbosch using a portable spectrophotometer (Aqualitic COD-reactor®, Dortmund) and the appropriate test kits (COD, CSB, 0-15000 mgL^{-1}). The samples were also analysed by a commercial laboratory (Bemlab, Strand) for pH, EC, P ($H_2PO_4^-$), K^+, Na^+, Cl^-, HCO_3^-, SO_4^{2-} and Fe^2 according to methods described by Clesceri *et al.* (1998). The Ca^{2+}, Mg^{2+}, K^+ and Na^+ in the water were determined by inductively coupled plasma optical emission spectrometry (ICP-OES) using a spectrometer (Perkin-Elmer Optima 7300 DV, Waltham, Massachusetts). The cation concentrations in mg/L^{-1} were converted to $meq.L^{-1}$ in order to calculate the sodium adsorption ratio (SAR) as follows:

$$SAR = Na^+ \div [(Ca^{2+} + Mg^{2+}) \div 2]^{0.5} \qquad \text{(Eq. 3.1)}$$

Total dissolved solids (TDS) was estimated by multiplying the EC of the water using a factor of 6.4 as proposed by the Department of Water Affairs and Forestry (1996).

3.2.7. Soil sampling and analysis

At both sites, soil samples were collected before wastewater monitoring began in March 2011. Following this, samples were collected in May before winter rainfall and in November after the winter rainfall during 2011, 2012 and 2013. Soil samples were collected in 0 - 10 cm, 10 - 20 cm, 20 - 30 cm, 30 - 60 cm and 60 - 90 cm depth increments. Physical limitations prevented the collection of soil samples deeper than 90 cm at both localities. All soil analyses were carried out by a commercial laboratory (Bemlab, Strand). Total organic C contents were determined using the method described by Walkley and Black (1934). The $pH_{(KCl)}$ was determined in a 1 M potassium chloride (KCl) suspension. The Ca^{2+}, Mg^{2+}, K^+ and Na^+ were extracted with 1 M ammonium acetate at pH 7. The cation concentrations in the extracts were determined by means of atomic emission using an optical emission spectrometer (Varian ICP-OES) at a commercial laboratory (BEMLAB, Strand). For this study, the cations will be referred to as extractable calcium (Ca^{2+}_{extr}), magnesium (Mg^{2+}_{extr}), potassium (K^{+}_{extr}) and sodium (Na^{+}_{extr}). The extractable potassium percentage (EPP') was calculated as follows:

$$EPP' = (K^{+}_{extr} \div S) \times 100 \qquad \text{(Eq. 3.2)}$$

where K^{+}_{extr} is the extractable potassium ($cmol^{(+)}/kg$) and S is the sum of basic cations ($cmol^{(+)}/kg$). The extractable sodium percentage (ESP') was calculated in the same way to obtain an indication of the sodicity status. Phosphorus was determined according to the Bray No. 2 method, *i.e.* extraction with 0.03 M NH_4F (ammonium-fluoride) in 0.01 M HCl (hydrochloric acid). The P concentration in the extract was determined by means of atomic emission as mentioned above. The soil's CEC was determined using 0.2 M ammonium acetate (pH=7 as extractant of exchangeable cations) method as described by The Non-affiliated Soil Analyses Work Committee (1990).

3.2.8 Statistical procedures

The experimental design was a randomised complete block with seven sampling times randomly replicated within each of three blocks. At each sampling time determinations were made at five depths intervals. Univariate analysis of variance was performed, for each depth separately, on all variables accessed using GLM (General Linear Models) Procedure of SAS statistical software (Version 9.2; SAS Institute Inc., Cary, NC, USA). Values for different depths were also combined in a split-plot analysis of variance with depth as sub-plot factor (Snedecor, 1980). Shapiro-Wilk test was performed to test for normality (Shapiro, 1965). Student's t-least significant difference was calculated at the 5% level to compare treatment means (Ott, 1998). A probability level of 5% was considered significant for all significance tests.

3.3. RESULTS AND DISCUSSION

3.3.1. Winery near Rawsonville

3.3.1.1. Chemical composition of winery wastewater

Basic cations: It was evident that the wastewater contained high amounts of K^+ and Na^+ which could have a negative impact on the soil (Fig. 3.4A). On average, K^+ levels in the wastewater were substantially higher than the levels of Na^+. This indicated that the winery probably used more K^+ containing detergents than Na^+ based ones. The annual fluctuation in K^+ and Na^+ could not be related to specific seasonal activities in the winery, *e.g.* grape crushing or bottling. However, almost throughout the study period the Na^+ was higher than 70 mg/L, *i.e.* the upper threshold for unrestricted use for sprinkler irrigation (Ayers & Westcot, 1994). The levels of Ca^{2+} and Mg^{2+} in the wastewater were substantially lower than the monovalent ions (Fig. 3.4B). This was to be expected since chemicals containing Ca^{2+} and Mg^{2+} does not play a prominent role in winery processes. At these low levels the bivalent ions would not have any negative effects on soils or crops. However, the Ca^{2+} and Mg^{2+} could have some positive effect on the water quality by reducing the SAR.

SAR: In 2011, the winery wastewater SAR was frequently higher than 5, *i.e.* the legal limit for irrigation with wastewater as stipulated in the Department of Water Affairs (2013) General Authorization (Fig. 3.4C). During the remainder of the study period, the SAR was mostly equal to, or below the legal limit. It should be noted that the wastewater SAR did not follow a distinct annual pattern that could be related to specific activities in the winery.

EC: The winery wastewater EC was below the permissible limit of 2 dS/m, *i.e.* as stipulated in the Department of Water Affairs (2013) General Authorization for irrigation with wastewater, except for prominent spikes in January 2012 and June 2013 (Fig. 3.4D). Similar to the SAR, the EC did not follow a distinct annual pattern that could be related to specific winery activities.

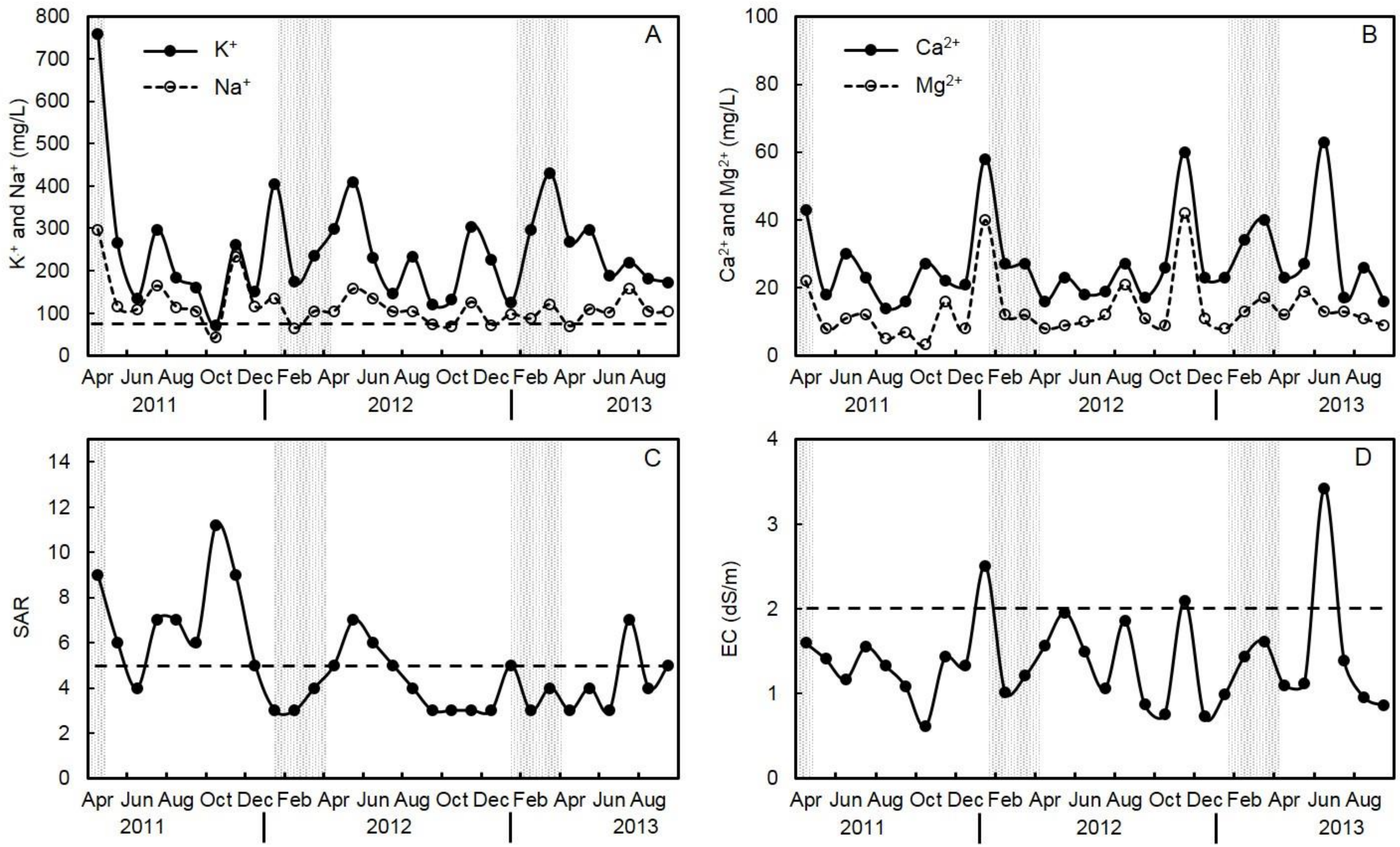

Figure 3.4. Temporal variation in (A) K^+ and Na^+, (B) Ca^{2+} and Mg^{2+}, (C) sodium adsorption ratio (SAR) and (D) electrical conductivity (EC) in wastewater from a winery near Rawsonville. Shaded columns indicate the harvest periods. Dashed lines indicate the Na^+, SAR and EC thresholds for irrigation water.

Anions: Similar to the cations, the variation in levels of HCO_3^-, as well as SO_4^{2-} and Cl^- could not be related to a specific activity in the winery (Fig. 3.5A & B). During February and March 2013, the level of Cl^- was above the recommended threshold of 150 mg/L for vineyard irrigation (Howell & Myburgh, 2013, and references therein) (Fig.3.5B).

Phosphorus: Since the levels of P were generally low throughout the study period (Fig. 3.5B), land application of the wastewater would not make a significant contribution to the P requirements of crops.

pH: With the exception of November and December 2011, the winery wastewater pH was generally equal to or less than 6, *i.e.* the lower limit for wastewater irrigation as stipulated in the Department of Water Affairs (2013) General Authorization (Fig. 3.5C). Annually, the pH tended to be higher in winter than during the harvest period. Since the pH was below the legal requirement for disposal through land application during these periods, it was not suitable for irrigation of crops. Based on the foregoing, the experiment plots were irrigated with acidic water throughout most of the study period.

COD: Throughout the study period, the winery wastewater COD was considerably higher than 400 mg/L, *i.e.* the upper limit for wastewater irrigation as stipulated in the Department of Water Affairs (2013) General Authorization (Fig. 3.5D). Therefore, the wastewater did not comply with the legislation for disposal through land application. Furthermore, the COD frequently exceeded 5000 mg/L, *i.e.* the threshold where wastewater may not be used for irrigation, or any other land application (Department of Water Affairs, 2013). Annually, the wastewater COD tended to peak during the harvest period (Fig. 3.5D). This confirmed that the crushing and wine making processes generated wastewater containing high levels of COD.

Iron: The fluctuation in levels of Fe could not be related to a specific seasonal activity in the winery (Fig. 3.6). The Fe levels were most of the time below the maximum acceptable water quality norm of 5 mg/L for continuous irrigation of grapevines (Howell & Myburgh, 2013 and references therein).

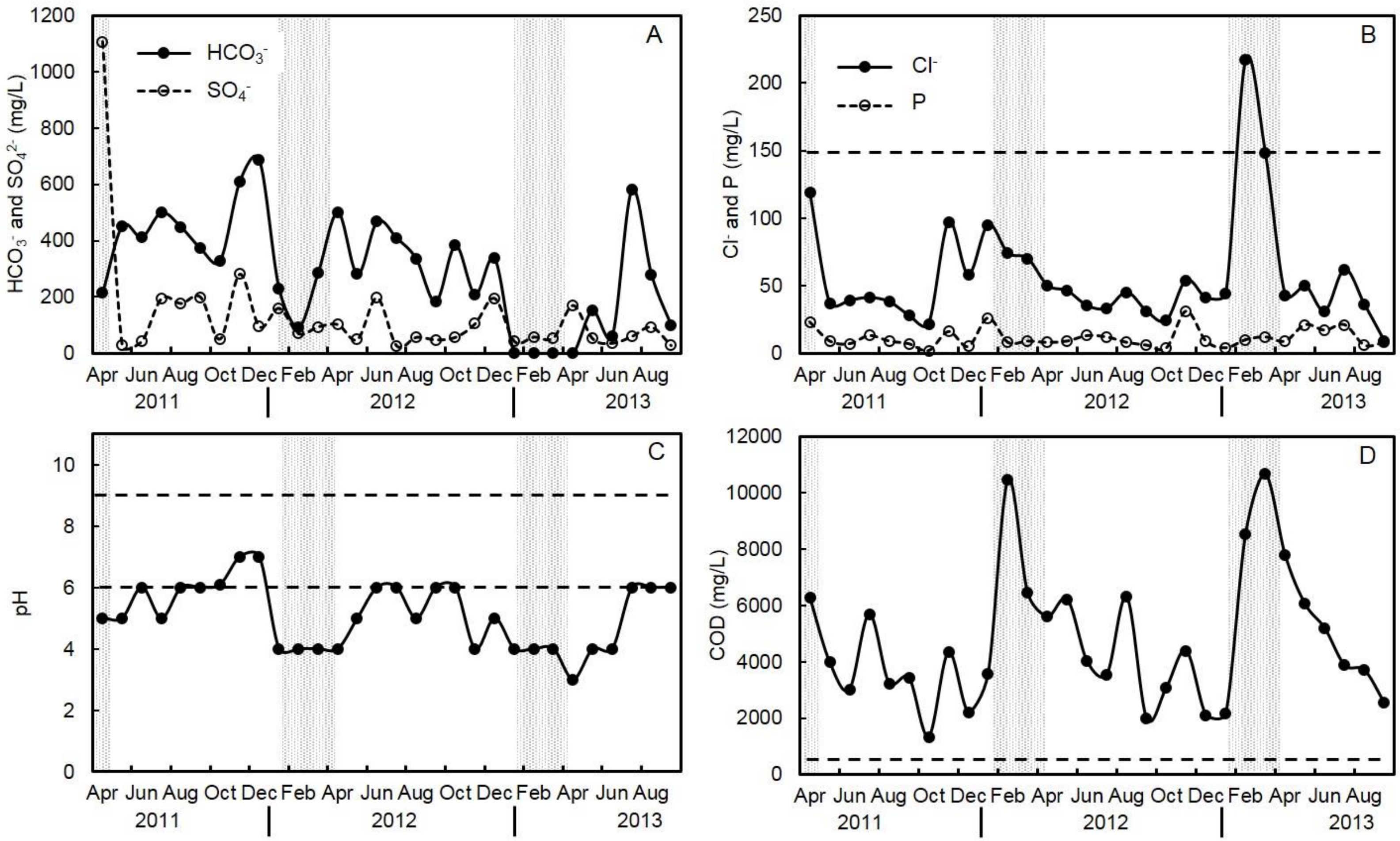

Figure 3.5. Temporal variation in (A) HCO_3^- and SO_4^{2-}, (B) Cl^- and P, (C) pH and (D) chemical oxygen demand (COD) in wastewater from a winery near Rawsonville. Shaded columns indicate the harvest periods. Dashed lines indicate Cl^-, pH and COD thresholds.

<u>TDS</u>: The fluctuation in levels of TDS could not be related to a specific seasonal activity in the winery (Fig 3.7). However, almost throughout the study period the TDS was higher than 450 mg/L, *i.e.* the upper threshold for unrestricted use for irrigation (Ayers & Westcot, 1994).

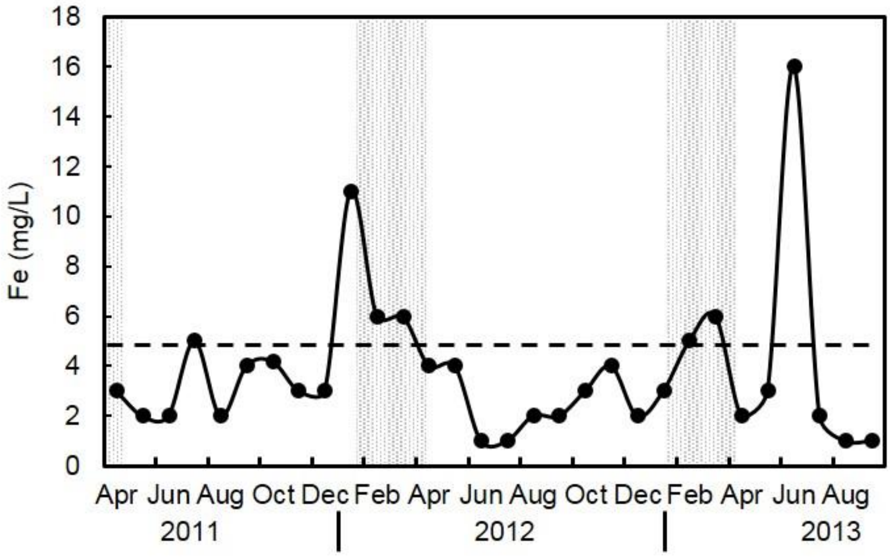

Figure 3.6. Temporal variation of the iron in winery wastewater used to irrigate an existing grazing paddock at a winery near Rawsonville. Shaded columns indicate the harvest periods. The dashed line indicates the maximum Fe^{2+} level for continuous irrigation.

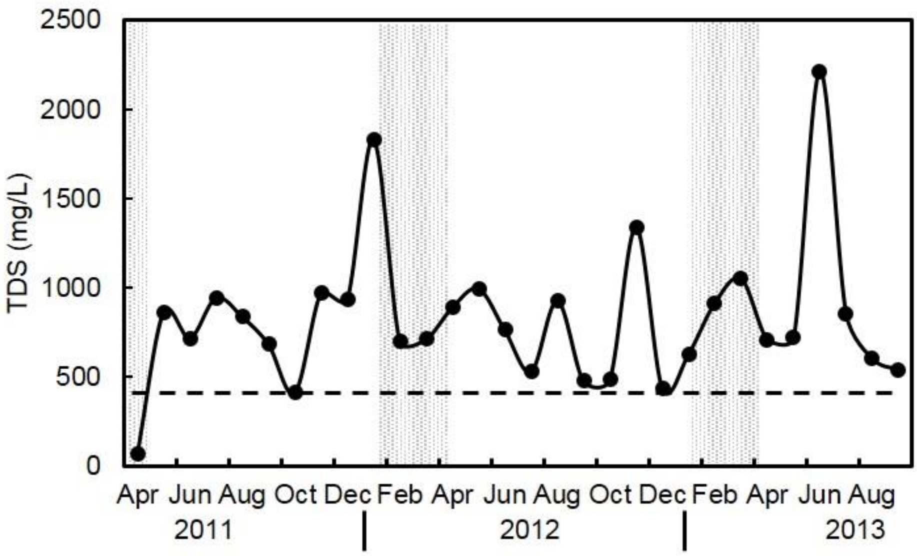

Figure 3.7. Temporal variation of the iron and total dissolved solids (TDS) in winery wastewater used to irrigate an existing grazing paddock at a winery near Rawsonville. Shaded columns indicate the harvest periods. The dashed line indicates the limit for unrestricted irrigation use.

3.3.1.2. Rainfall and volumes of wastewater applied

Mean monthly rainfall was typical for a Mediterranean climate (Fig. 3.8). However, it must be noted that the July rainfall was abnormally low in all the winters. Winter rainfall, *i.e.* from April to September, amounted to 380 mm, 420 mm and 685 mm in 2011, 2012 and 2013, respectively. As expected, wastewater irrigations were substantially higher in the harvest period, *i.e.* from February until April (Fig. 3.9). During the peak period, in March, *c.* 23 mm irrigation was applied per day. In December, the soil received only *c.* 3 mm wastewater per day. The irrigation volumes also increased from mid-winter to reach a second peak in August. Total irrigation applied during winter, *i.e.* from April to September, amounted to 1475 mm, 2600 mm and 3285 mm in 2011, 2012 and 2013, respectively. Based on the foregoing, the soil received the highest irrigation plus rainfall in the winter of 2013, followed by 2012 and then 2011.

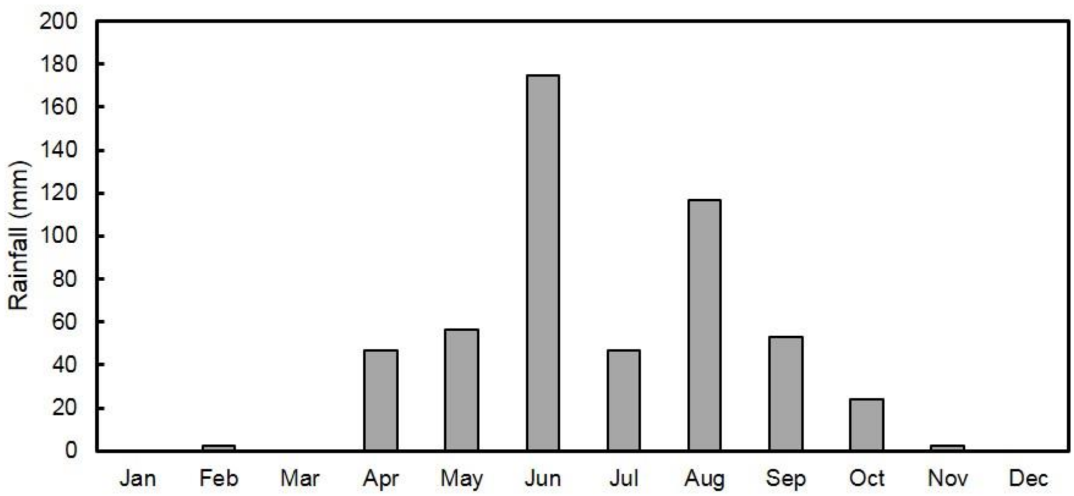

Figure 3.8. Mean monthly rainfall during the study period at the winery near Rawsonville.

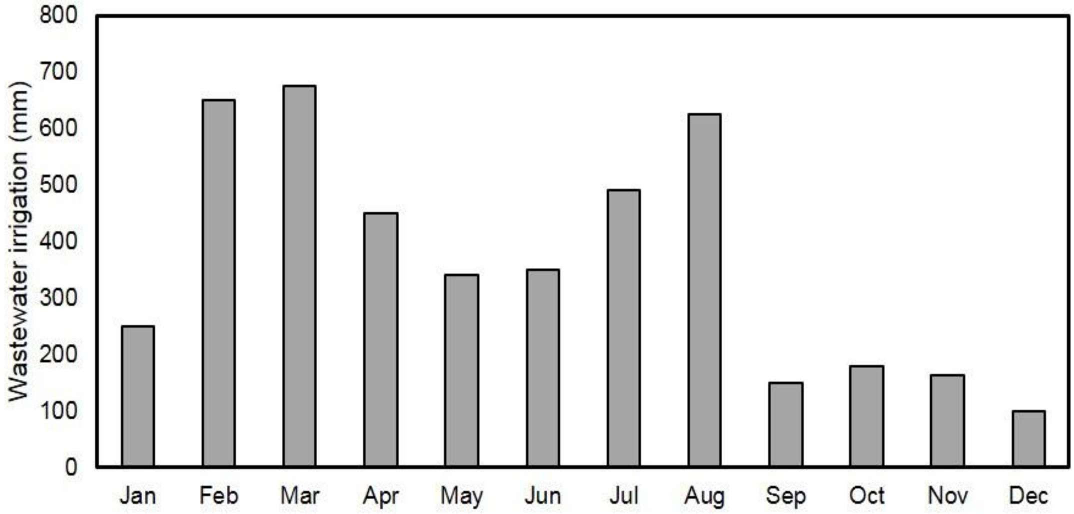

Figure 3.9. Mean monthly wastewater applied during the study period at the winery near Rawsonville.

Since wastewater was applied to a poorly drained soil on level land, the soil became totally waterlogged. Visual observations revealed that the water actually ponded on the soil after the irrigation was applied, particularly in winter (Fig. 3.10A). A previous study showed that a similar poorly drained, grey, sandy soil developed a water table deep in the profile (Mulidzi *et al*., 2002). This particular soil was on a slight slope which allowed lateral drainage, thereby preventing the entire profile from becoming waterlogged.

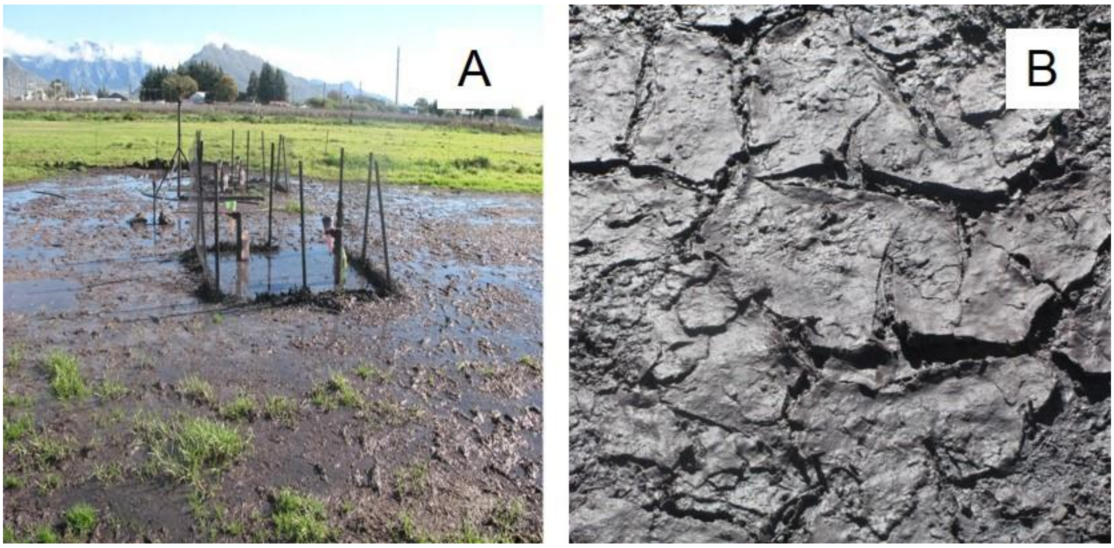

Figure 3.10. Waterlogging upon irrigation with wastewater caused (A) ponding and die-back of the grass, as well as (B) accumulation of organic matter on the surface of the Longlands soil form at a winery near Rawsonville.

Due to the waterlogging, part of the water soluble organic fraction of the wastewater accumulated in the topsoil and in the ponded water on the soil surface. The organic matter probably underwent anaerobic decomposition, which caused bad odours in the vicinity of the ponded water (Fig. 3.10B). This is in contrast with a previous study which showed that the anaerobic decomposition of the organic matter occurred deep in the soil profile (Mulidzi *et al*., 2002). Prior to soil sampling, no obnoxious odours were noticed. Obnoxious odours were only noticed when soil samples were collected from the deepest layers by means of an auger. Therefore, the source of the obnoxious odours seemed to be contained in the deepest soil layers. Given the fact that the soil was not entirely waterlogged, the organic matter probably leached into the subsoil.

The application of winery wastewater at the Rawsonville winery resulted in dieback of the grass on the irrigated area after only one month (Fig. 3.10A). This might have been the result of oxygen depletion in the topsoil due to the high level of COD in the wastewater. Most wineries that dispose of their wastewater through land application do not measure how much wastewater they are applying and their strategy is to irrigate an area until the plants die off and then move the sprinkler. The plants normally recover after three months. On areas where irrigation was conducted on well drained soils on a sloping areas, it was observed that the grass did not die back which was associated to the deeper percolation of water containing high levels of organic matter (Mulidzi, 2001).

3.3.1.3. Soil chemical status

3.3.1.3.1. Initial soil chemical status

After continuous irrigation with winery wastewater for 15 years, the soil was acidic throughout the profile, *i.e.* the $pH_{(KCl)}$ was less than 4.5 (Table 3.3). The soil Bray IIP was high in all soil layers, *i.e.* more than 20 mg/kg which is considered to be the norm for sandy soils (Conradie, 1994). The basic cations declined with depth. By far the highest concentration of all cations occurred in the 0-10 cm layer (Table 3.3). These levels were relatively high for sandy soils. This suggested that the sludge probably had a high CEC. The Ca_{extr} was the dominant cation, whereas Na_{extr} was the lowest throughout the profile. The EPP' was relatively high in the deepest soil layers (Table 3.3). In contrast, the ESP' was highest near the soil surface.

Table 3.3. Chemical status of the Longlands soil that was irrigated with winery wastewater over a 15 year period near Rawsonville before the study began.

Depth (cm)	$pH_{(KCl)}$	P (mg/kg)	Basic extractable cations ($cmol_c.kg^{-1}$)				EPP' (%)	ESP' (%)
			Na^+_{extr}	K^+_{extr}	Ca^{2+}_{extr}	Mg^{2+}_{extr}		
10	4.1	270	1.5	2.2	8.4	2.5	15.3	10.4
20	4.3	209	0.5	0.9	4.9	0.7	12.9	6.7
30	4.6	208	0.4	0.7	3.9	0.6	11.9	7.1
60	4.6	255	0.1	0.5	0.9	0.2	26.9	7.0
90	4.6	264	0.1	0.5	0.9	0.2	28.6	7.9

3.3.1.3.2. Soil chemical status during the study period

Organic carbon: Soil organic C in the 0-10 and 10-20 cm layers was substantially higher than 2% (Fig. 3.11), which is relatively high for soils of the Western Cape wine regions. This indicated that organic matter applied *via* the winery wastewater had accumulated in the layers near the soil surface. Except for May 2012, when the organic carbon in the 0-10 cm layer showed a peak, it tended to remain constant over the two-and-a-half-year period. The sludge observed at the surface probably contributed to the exceptionally high organic carbon in the 0-10 cm layer. Furthermore, it must be noted that the organic carbon at the end of the period was comparable to the initial level at the beginning of the study in March 2011. The organic carbon in the 10-20 cm layer showed an increase until May 2012. This suggested that some of the organic matter had leached into the soil by the high irrigation volume. The organic carbon in the 10-20 cm layer tended to remain constant from May 2012 until the end of the study period. The organic carbon in the 10-20 cm layer tended to decline following November 2011. At this stage there is no explanation for this trend. Since the organic carbon in the deeper layers remained almost unchanged, it is unlikely that organic carbon could have leached from the 20-30 cm layer into these layers.

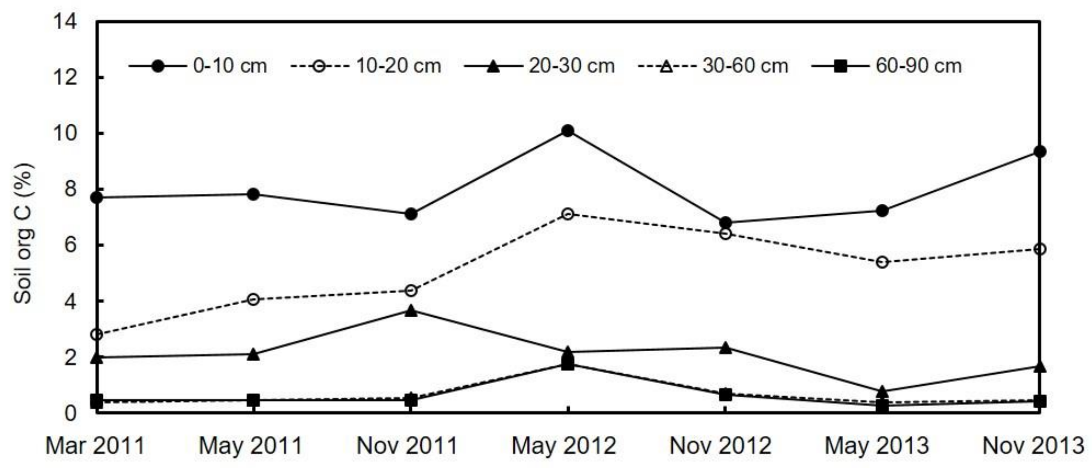

Figure 3.11. Temporal variation in soil organic C where winery wastewater was applied to a Longlands soil near Rawsonville.

<u>Potassium</u>: Substantial volumes of wastewater was applied between the different sampling times, particularly from November until May (Fig. 3.12). As expected, the contribution of rainfall to the total volume of water that the soil received was higher during winter than in summer. It was evident that application of winery wastewater increased the K^{+}_{extr} levels in the 0-10 cm layer, and to some extent in the 10-20 cm layer, at the end of the harvest periods (Fig. 3.13). Despite the seasonal fluctuations, K^{+}_{extr} steadily increased over the three years in the first two soil layers compared to the levels at the beginning of the study. After three years of wastewater application there was no significant increase in K^{+}_{extr} levels deeper than 20 cm depth (Fig. 3.13).

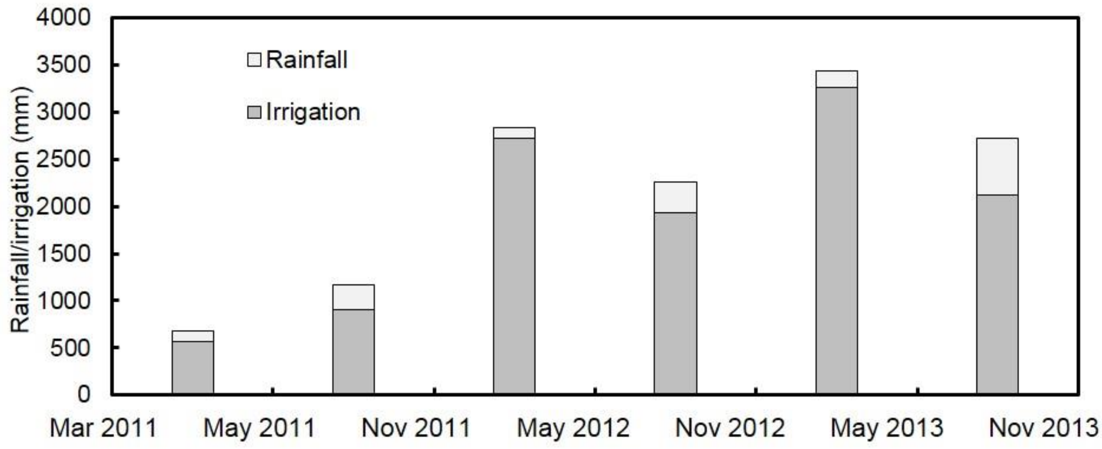

Figure 3.12. Temporal variation in rainfall and winery wastewater irrigation as measured near Rawsonville.

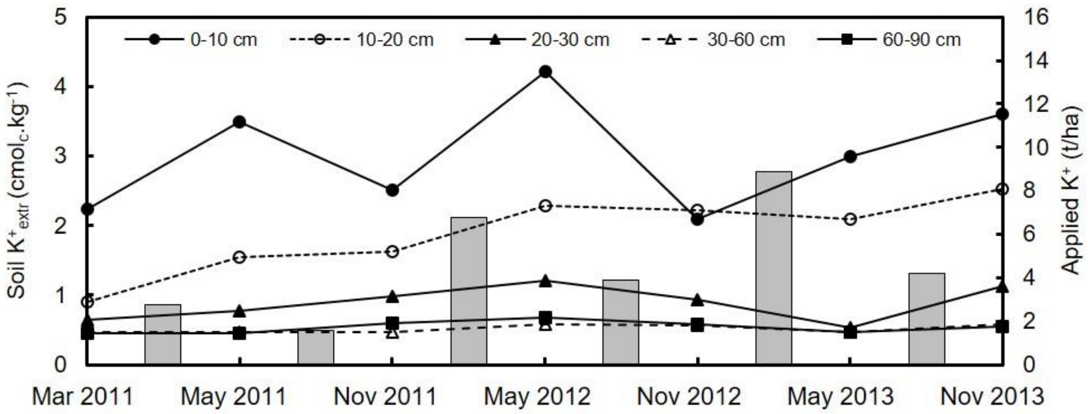

Figure 3.13. Temporal variation in soil extractable K^+ and the amount of K^+ applied *via* winery wastewater to a Longlands soil near Rawsonville. Vertical columns indicate applied K^+.

The sludge deposits on the soil surface (Fig. 3.10B) probably retained high levels of K^+ by the end of the harvest period. During winter, when less K^+ was applied (Fig. 3.13), some of the K^+ probably leached from the sludge, causing the lower levels in the 0-10 cm layer. Since there was little change in K^+ levels with depth throughout the profile, it suggested that most of the applied K^+ was leached beyond 90 cm. In fact, seasonal soil K^+ balances showed that substantial amounts of K^+ was leached (Table 3.4). Furthermore, the cumulative leached K^+ was linearly related to the cumulative irrigation plus rainfall (Fig. 3.14). Due to the low clay content of the soil (Table 3.1), the exchange complex could not retain large amounts of K^+. Therefore, leaching of K^+ beyond 90 cm was not inhibited. Although, leaching of K^+ from sandy or coarse textured soils during winter rainfall reduces the risk of accumulation and clay dispersion, it increases environmental risks such as groundwater recharge and/or lateral flow into other fresh water resources.

Table 3.4. Soil extractable K^+ balances for selected periods in the 0-90 cm depth of a sandy Longlands soil that was irrigated with winery wastewater near Rawsonville.

Period	Soil K^+ (kg/ha)		Applied K^+	K^+ loss	Leached K^+
	Beginning	End	(kg/ha)	(kg/ha)	(%)
Mar 11 - May 11	3978	5909	2768	837	30
May 11 - Nov 11	5909	4914	1561	2556	164
Nov 11 - May 12	4914	6786	6760	4888	72
May 12 - Nov 12	6786	5148	3894	5532	142
Nov 12 - May 13	5148	5031	8879	8996	101
May 13 - Nov 13	5031	6318	4186	2899	69

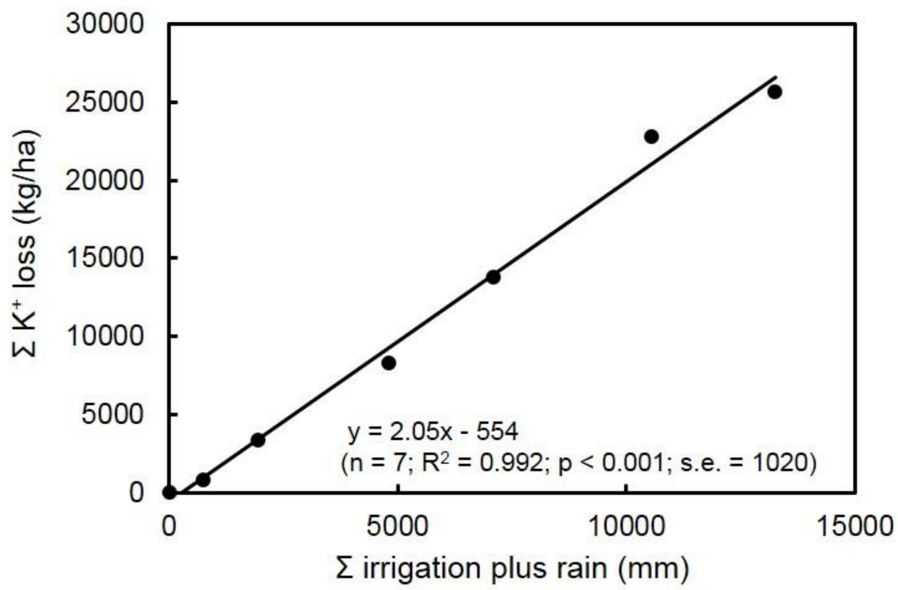

Figure 3.14. Effect of cumulative (Σ) rainfall + irrigation - evapotranspiration on cumulative K^+ losses beyond 90 cm depth where a Longlands soil was irrigated with winery wastewater near Rawsonville.

Sodium: Similar to K^+_{extr}, irrigation with winery wastewater increased the Na^+_{extr} levels in the 0-10 cm and in the 10-20 cm layers, at the end of the harvest periods (Fig. 3.15). In May 2012, the Na^+_{extr} was also slightly higher in the 20-30 cm layer compared to the rest of the study period. Despite the seasonal fluctuations, Na^+_{extr} tended to increase slightly over the two-and-a-half-year study period in the first two soil layers compared to the levels at the beginning of the study. At the end of the study period, there was no increase in Na^+_{extr} deeper than 20 cm depth (Fig. 3.15).

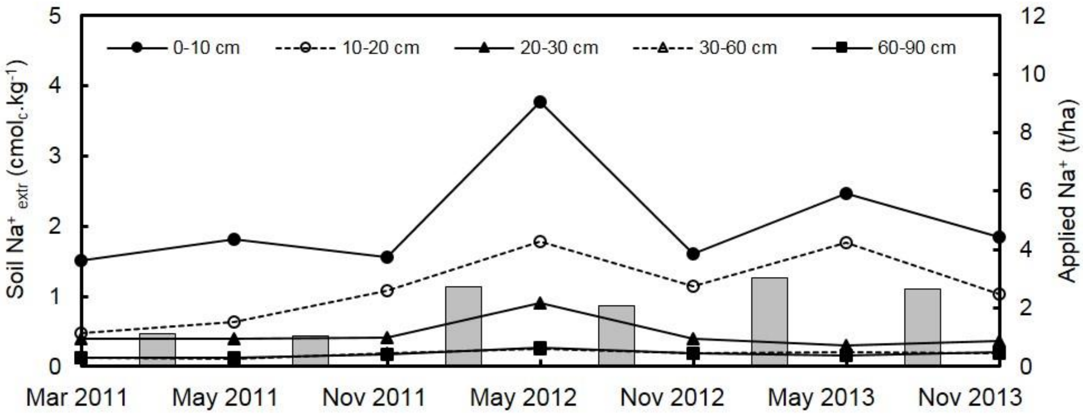

Figure 3.15. Temporal variation in soil extractable Na^+_{extr} and amount of Na^+ applied *via* wastewater to a Longlands soil near Rawsonville. Vertical columns indicate applied Na^+.

The sludge deposits on the soil surface probably also retained high levels of Na^+ by the end of the harvest periods. During winter, when less Na^+ was applied (Fig. 3.15), some of the Na^+ probably leached from the sludge, causing the lower levels in the 0-10 cm and 10-20 cm layers. Since there was little change in Na^+_{extr} levels with depth throughout the profile, it suggested that most of the applied Na^+ was leached beyond 90 cm. Seasonal soil Na^+ balances confirmed that substantial amounts of Na^+ was leached (Table 3.5). Furthermore, the cumulative leached Na^+ was also linearly related to the cumulative irrigation plus rainfall (Fig. 3.16). Similar to K^+, the low clay content of the soil could probably not retain large amounts of Na^+. Therefore, leaching of Na^+ beyond 90 cm was also not inhibited. Although, leaching of Na^+ from sandy or coarse textured soils during winter rainfall also reduces the risk of accumulation and dispersion, it poses the same environmental risks as the large amounts of K^+ that was leached from the soil.

High concentrations of Na^+ in soil due to winery wastewater application can reduce soil aggregate stability (Laurenson & Houlbrooke, 2012). When Na^+ is the predominant adsorbed cation, the clay disperses. When the soil is wet, puddling reduces permeability, and when it is dry a hard impermeable crust forms. This suggested that high levels of Na^+ in the sludge could have caused the ponding on the soil surface (Fig. 3.10). However, it does not rule out the possibility that the high organic matter content could have clogged the soil pores near the surface, which reduced infiltration.

Table 3.5. Soil extractable Na^+ balances for selected periods in the 0-90 cm depth increment of a sandy Longlands soil that was irrigated with winery wastewater near Rawsonville.

Period	Soil Na^+ (kg/ha)		Applied Na^+ (kg/ha)	Na^+ loss (kg/ha)	Leached Na^+ (%)
	Beginning	End			
Mar 11 - May 11	1035	1173	1117	979	88
May 11 - Nov 11	1173	1484	1060	749	71
Nov 11 - May 12	1484	2664	2734	1354	50
May 12 - Nov 12	2864	1484	2076	3456	167
Nov 12 - May 13	1484	2001	3046	2529	83
May 13 - Nov 13	2001	1553	2658	3106	117

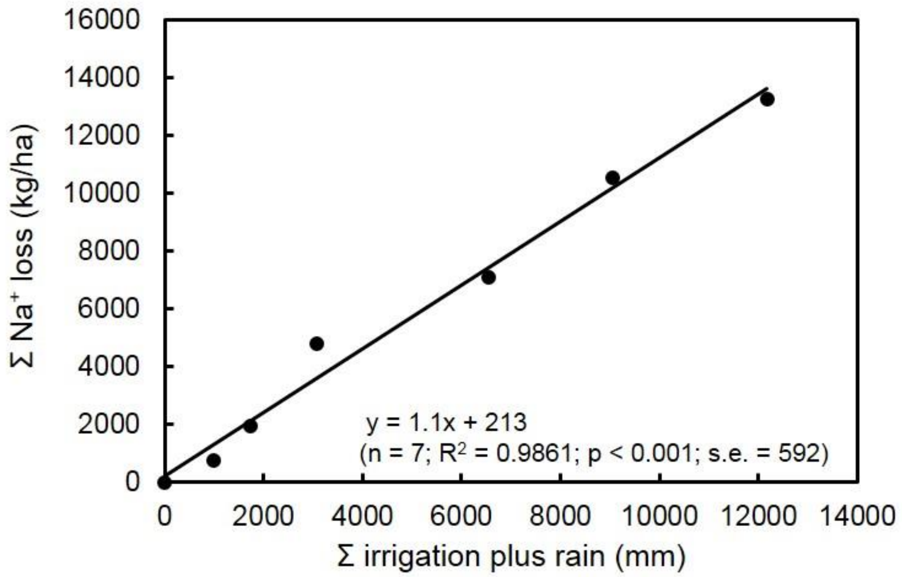

Figure 3.16. Effect of cumulative (Σ) irrigation plus rainfall on cumulative Na^+ losses beyond 90 cm depth where a Longlands soil was irrigated with winery wastewater near Rawsonville.

<u>Calcium</u>: The Ca_{extr} in the 0-10 cm and 10-20 cm layers, and to a lesser extent in the 20-30 cm layer, tended to increase at the end of the harvest period (Fig. 3.17). This was followed by a decline during winter. It is interesting to note that the seasonal variation in Ca^{2+}_{extr} occurred in the 30-60 cm layer although the concentrations were considerably lower compared to the topsoil. A previous study showed that continuous application of winery wastewater high in K^+ and Na^+ could cause the soil exchange sites to be dominated by monovalent ions, thereby pushing bivalent ions such as Ca^{2+} and Mg^{2+} out of the exchange complex (Mosse *et al.,* 2011). Consequently, the bivalent cations will be leached from the soil. However, the Ca_{extr} in the deeper layers remained constant throughout the study period under the prevailing conditions. Although Ca^{2+} levels were generally low in the winery wastewater, it seemed that higher applications during the harvest period reflected in the Ca_{extr}. Since the applied Ca^{2+} was substantially lower than amounts of K^+ and Na^+, it is unlikely that the Ca^{2+} would affect the EPP' or ESP' significantly. Therefore, the bivalent cations will probably not counter structural problems caused by high amounts of K^+ and Na^+ from the wastewater when applied to the soil.

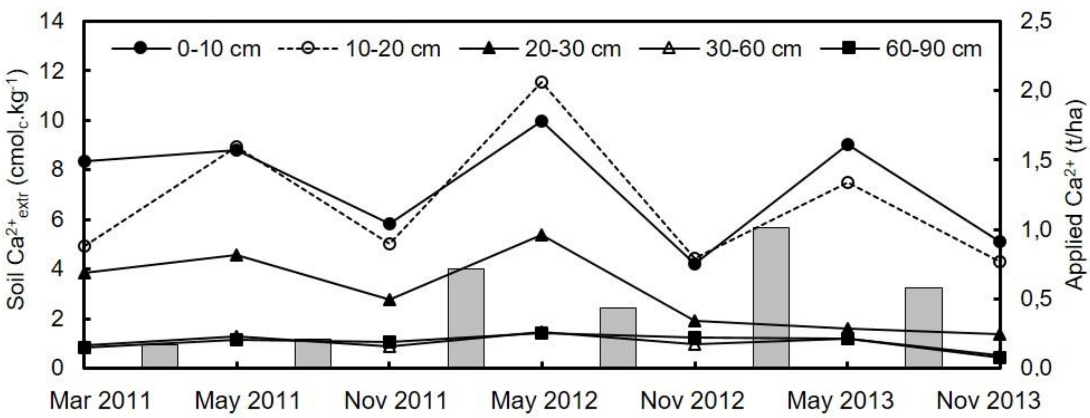

Figure 3.17. Temporal variation in soil extractable Ca^{2+} and amounts of Ca^{2+} applied where wastewater was applied to a Longlands soil near Rawsonville. Vertical columns indicate applied Ca^{2+}.

<u>Magnesium</u>: The Mg^{2+}_{extr} in the 0-10 cm, and to a lesser extent in the 10-20 cm layer, showed the same seasonal fluctuation as the Ca^{2+}_{extr} (Fig. 3.18). The Mg_{extr} in the deeper layers remained more or less constant throughout the study period. Although Mg^+ levels were generally low in the winery wastewater, it seemed that higher applications during the harvest period also reflected in the

Mg_{extr}. Similar to Ca^{2+}, the low levels of Mg^{2+} are unlikely to counter the negative effects of high K^{+} and Na^{+} applications on EPP' or ESP', and consequently on soil physical conditions.

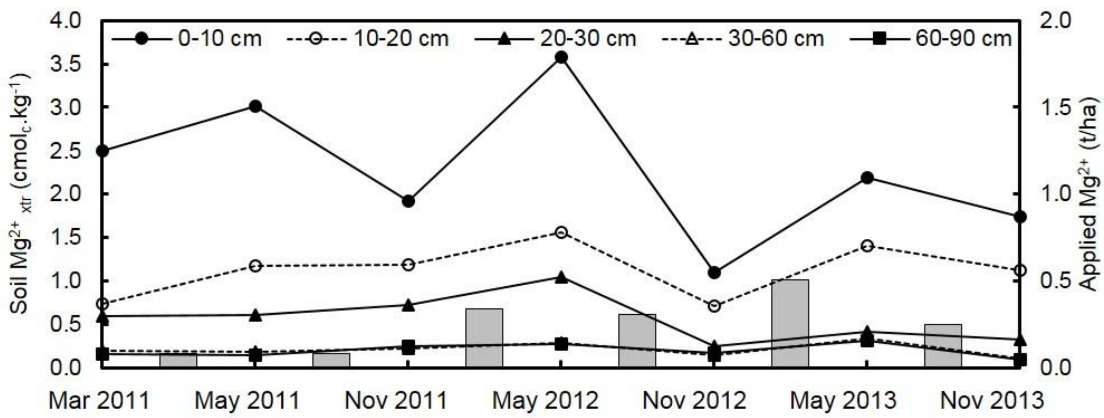

Figure 3.18. Temporal variation in soil extractable Mg^{2+} where wastewater was applied to a Longlands soil near Rawsonville. Vertical columns indicate applied Mg^{2+}.

EPP': With the exception of the 0-10 cm layer, the EPP' tended to be lower at the end of the harvest period, followed by an increase during winter (Fig. 3.19). This result is somewhat unexpected, since the higher EPP' did not correspond with the higher K^{+} applications which caused higher K^{+}_{extr} in the soil (Fig. 3.13). Although substantially more K^{+} than Ca^{2+} was applied *via* the wastewater, Ca^{2+} was the dominant cation in all the soil layers except in November 2013 when the Ca^{2+}_{extr} levels were comparable to the other extractable cations in the deeper layers (Fig. 3.20). The source of the Ca^{2+} was probably lime that was added to the wastewater in order to increase the pH as part of the wastewater treatment carried out by the winery. Routine use of Ca^{2+} amendments including, yet not restricted to, lime, gypsum and calcium nitrate either added directly to wastewater or to soils will enable Ca^{2+} exchange and displacement of Na+ and K+. Winter application of Ca^{2+} amendments will ensure its percolation down the soil profile thereby ensuring good distribution of Ca^{2+} (Laurenson & Houlbrooke,2011). Quantification of this practice was beyond the scope of the study. In November 2013, the winery probably reduced, or stopped the lime application which caused the low soil Ca^{2+}_{extr}. Based on the foregoing, it seemed that high levels of Ca^{2+}_{extr} at the end of the harvest dominated the exchange complex to such an extent that the EPP' was reduced compared to the winter when the Ca^{2+}_{extr} was lower. The high EPP' in November 2013 was due to the low Ca^{2+}_{extr}. These results also suggested that the large amounts of applied K^{+} *via* the winery wastewater was not preferentially absorbed onto the exchange sites.

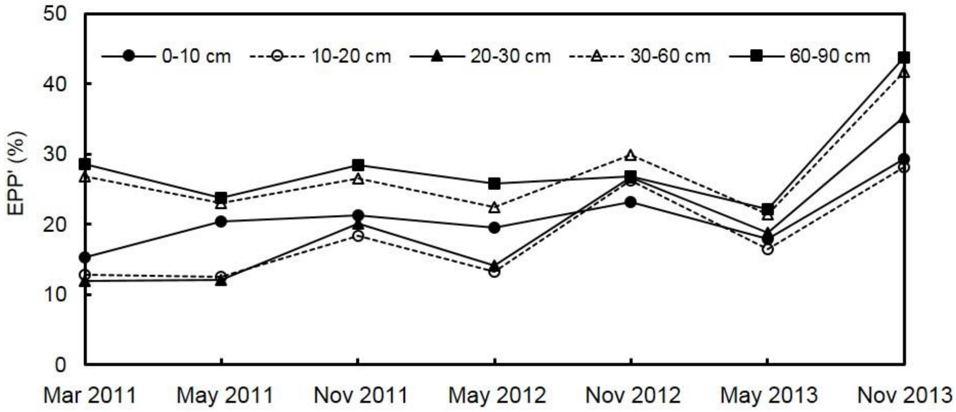

Figure 3.19. Temporal variation in soil EPP' where wastewater was applied to a Longlands soil near Rawsonville.

<u>ESP'</u>: Although the Na^{+}_{extr} showed some seasonal fluctuations (Fig. 3.15), it did not reflect in the ESP' (Fig. 3.21). The lack of seasonal fluctuations in ESP' was probably due to the dominance of Ca^{2+}_{extr}, and to some extent K^{+}_{extr}. It was previously reported that the adsorption of Na^{+} on soils similar to the Longlands soil was reduced by the presence of high levels of K^{+} after winery wastewater irrigation (Mulidzi *et al.*, 2016). High soil ESP' increases the risk of soil physical properties to deteriorate through clay dispersion which will lead to structural breakdown and blockage of soil pores and reduced soil permeability (Bond, 1998). However, since the ESP' was relatively low, it would probably not have caused serious soil physical deterioration.

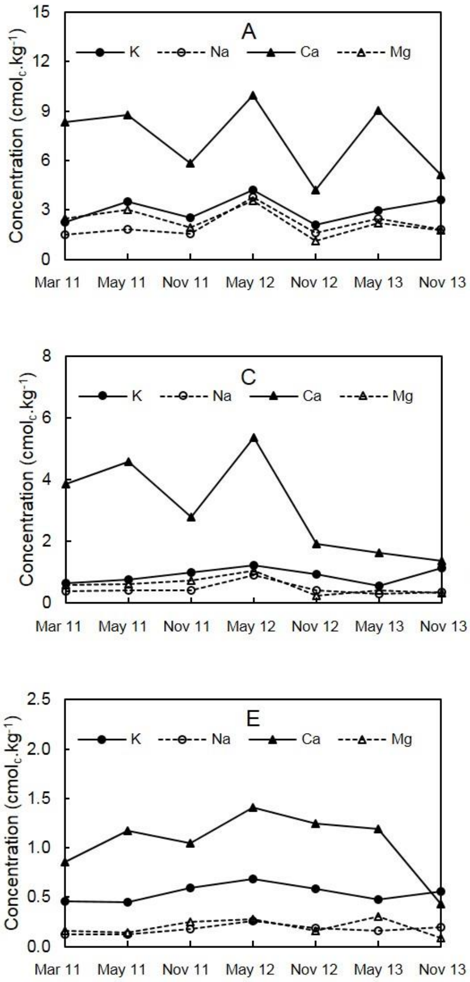

Figure 3.20. Temporal variation of the extractable cations in the (A) 0-10, (B) 10-20, (C) 20-30, (D) 30-60 and (E) 60-90 cm soil layers where winery wastewater was applied to a Longlands soil near Rawsonville.

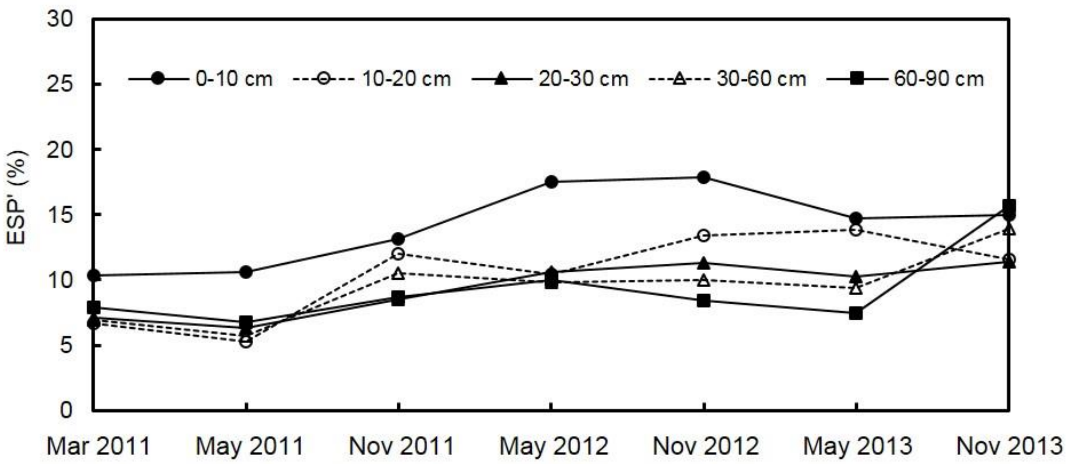

Figure 3.21. Temporal variation in soil ESP' where wastewater was applied to a Longlands soil near Rawsonville.

<u>EC_e</u>: The salt content remained fairly constant to a depth of 60 cm until May 2012, during which time the EC_e in the 60-90 cm layer tended to incline steadily (Fig. 3.22). Following the winter of 2012, EC_e in the deepest two soil layers declined. A similar trend also occurred in the winter of 2013. In fact, EC_e in all layers tended to be lower following May 2013. These results indicated that the high irrigation plus rainfall must have leached some of the salts applied *via* the winery wastewater irrigation beyond 90 cm depth, particularly in the last two winters.

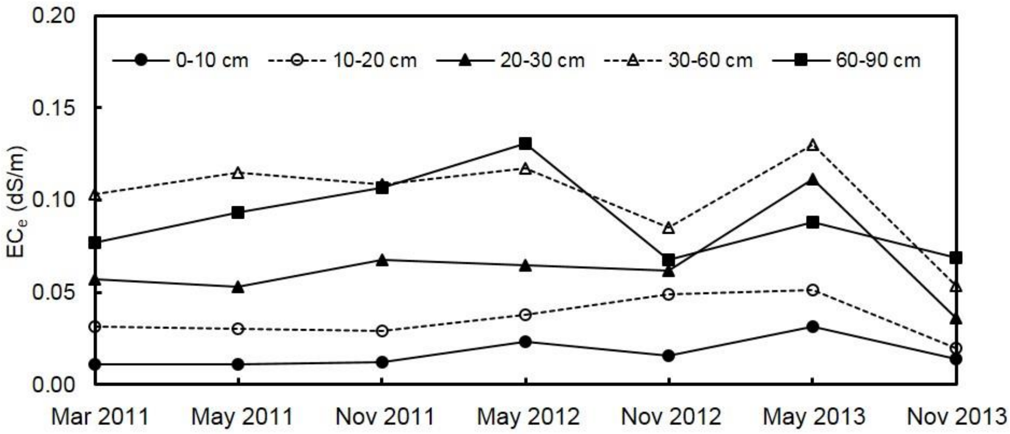

Figure 3.22. Temporal variation in electrical conductivity of the saturated soil paste (EC_e) where wastewater was applied to a Longlands soil near Rawsonville.

<u>$pH_{(KCl)}$</u>: Irrigation with winery wastewater slightly increased the soil $pH_{(KCl)}$ until May 2012 (Fig. 3.23). In November 2012 the soil $pH_{(KCl)}$ showed a decrease, and tended to remain constant until November 2013. Variation in soil $pH_{(KCl)}$ was not related to variation in monovalent cations (data not shown). However, addition of organic acids from winery wastewater could be associated with the decrease of soil pH due to H^+ dissociation from carboxyl functional groups (Rukshana *et al*., 2012). While the soil pH increase could be associated with high concentration of total alkalinity in wastewater that contains bicarbonate ions, as well as deprotonated organic acids, the charge of these ions are countered by cations. When applied to soils it increases the pH due to anion hydrolysis reactions and decarboxylation (Li *et al*., 2008). It is important to note that the soil was too acidic for viticulture, *i.e.* pH less than 5.5 (Conradie, 1994).

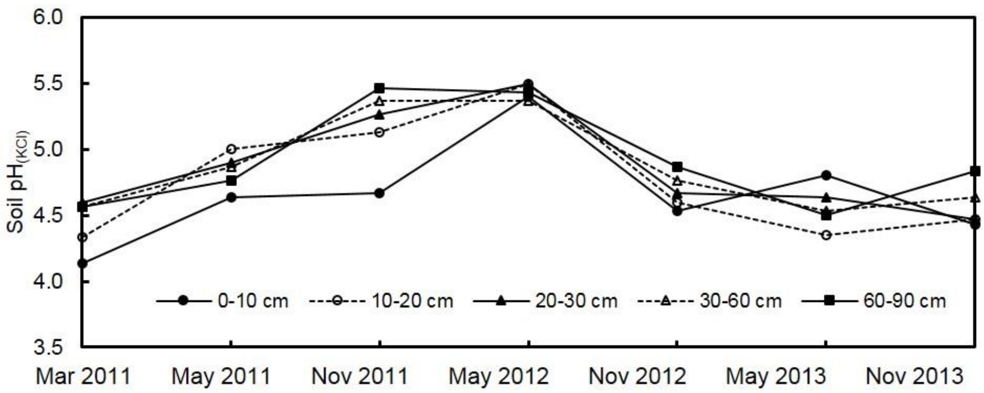

Figure 3.23. Temporal variation in soil pH$_{(KCl)}$ where wastewater was applied to a Longlands soil near Rawsonville.

<u>Phosphorus</u>: The soil P fluctuations appeared to be erratic (Fig. 3.24). At certain times, the P in the topsoil tended to increase, whereas the subsoil P tended to decline and *vice versa*. Therefore, it seemed that leaching of P into the subsoil occurred, which coincided with P losses from the topsoil. This was illustrated more clearly when the means for the topsoil (0-30 cm depth) and subsoil (30-90 cm depth) were plotted over time (Fig. 3.25). It seemed that the increase in subsoil P lagged behind P increases in the topsoil up till November 2012. Following this, top- and subsoil fluctuations coincided until November 2013. The high rainfall and irrigation before May 2013 probably caused leaching of P throughout the soil profile. However, this does not rule out the possibility that the low pH reduced the solubility of the P.

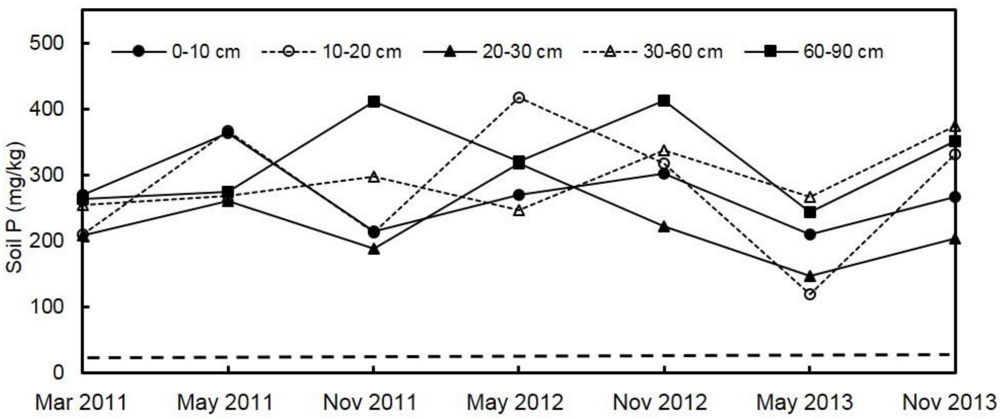

Figure 3.24. Temporal variation in soil P where wastewater was applied to a Longlands soil near Rawsonville. Dashed line indicate the proposed P norm for grapevines (Conradie, 1994).

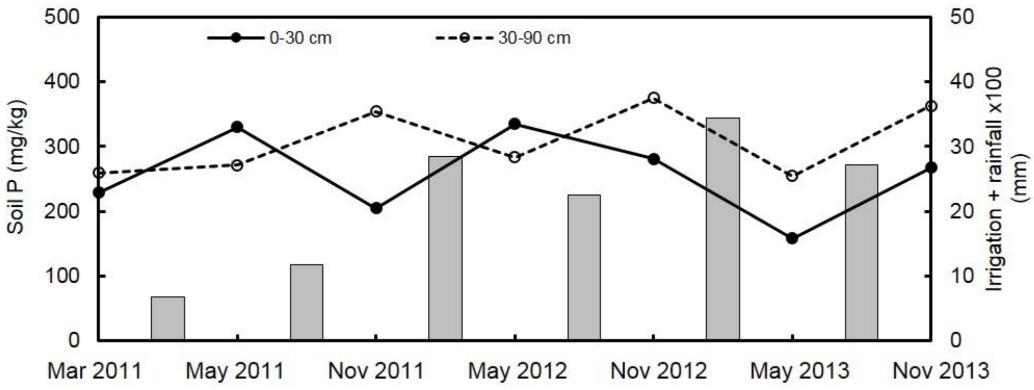

Figure 3.25. Temporal P variation in the topsoil (0-30 cm) and subsoil (30-90 cm), as well as irrigation plus rainfall where wastewater was applied to a Longlands soil near Rawsonville. Vertical columns indicate irrigation plus rainfall.

The soil P content was substantially higher than the minimum requirement for vineyards (Conradie, 1994) (Fig. 3.24). It must be noted that leaching of high levels of P into groundwater, as well as other fresh water sources close to the winery, could cause serious environmental problems, *e.g.* eutrophication. The leaching of P poses a very serious risk to the nearby water streams. Due to the sandy nature of the soil, *i.e.* 3.3% clay, and low Fe content, it does not have adequate P adsorbing capacity (Samadi, 2006). This would increase the risk of leaching excessive P from the soil.

3.3.2. Winery near Stellenbosch

3.3.2.1. Chemical composition of winery wastewater

<u>Basic cations</u>: It was evident that the wastewater contained high amounts of K^+, but relatively low levels of Na^+ (Fig. 3.26A). This indicated that the winery probably used more K^+ containing detergents than Na^+-based ones. Most of the time, the Na^+ was less than 70 mgL^{-1}, *i.e.* the upper threshold for unrestricted use with sprinkler irrigation (Ayers & Westcot, 1994). The annual fluctuation in K^+ and Na^+ could not be related to specific seasonal activities in the winery, *e.g.* grape crushing or bottling. The levels Ca^{2+} and Mg^{2+} in the wastewater were substantially lower than the monovalent ions (Fig. 3.26B). This was to be expected since chemicals containing Ca^{2+} and Mg^{2+} do not play a prominent role in winery processes. At these low levels the bivalent ions would not have any negative effects on soils or crops. However, the Ca^{2+} and Mg^{2+} could have some positive effect on the water quality by reducing the SAR of the wastewater.

<u>SAR</u>: Except in April and May 2001 (Fig. 3.26C), the wastewater SAR was well below 5, *i.e.* the legal limit as stipulated in the Department of Water Affairs (2013) General Authorization. This indicated that sodic soil conditions were unlikely to develop under the prevailing conditions. Similar to the Na^+, the wastewater SAR did follow a distinct annual pattern that could be related to specific activities in the winery.

<u>EC</u>: Although the EC of the winery wastewater was initially high (Fig. 3.26D), it gradually declined and from January 2012 until the end of the study period it was below, or equal to the permissible limit of 2 dS/m, *i.e.* the legal limit as stipulated in the Department of Water Affairs (2013) General Authorization. This indicated that saline soil conditions were unlikely to develop under the prevailing conditions. It should be noted that the EC did not follow a distinct annual pattern that could be related to specific activities in the winery.

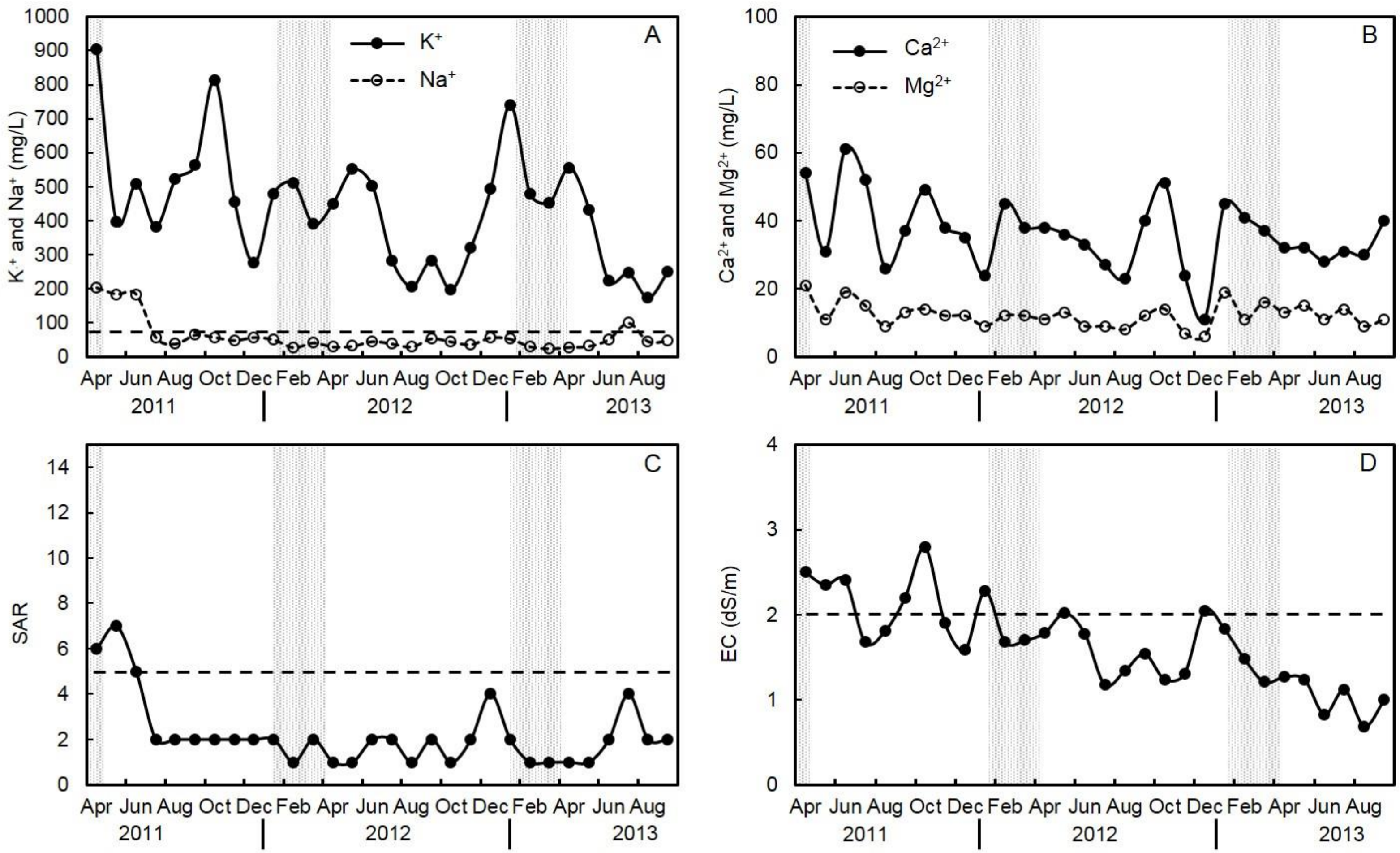

Figure 3.26. Temporal variation in (A) K^+ and Na^+, (B) Ca^{2+} and Mg^{2+}, (C) sodium adsorption ratio (SAR) and (D) electrical conductivity (EC) in wastewater from the Stellenbosch winery. Shaded columns indicate the harvest periods. Dashed lines indicate the Na^+, SAR and EC thresholds.

<u>Anions</u>: The level of HCO_3^- in the wastewater general tended to decline over the study period (Fig. 3.27A). However, the HCO_3^- content was relatively low during the harvest periods. Although irrigation with water containing high levels of HCO_3^- could affect soils, plants and irrigation equipment, there are no guidelines available (Howell & Myburgh, 2013 and references therein). Given the high levels in the winery wastewater (Fig. 3.27A), negative effects could be expected over time if the water is used for irrigation. The level of SO_4^{2-} in the wastewater was substantially lower than the HCO_3^- (Fig. 3.27A). Except for some spikes following the harvest period in 2013, the variation in SO_4^{2-} could not be related to a specific activity in the winery. Unlike the HCO_3^-, the Cl^- tended to increase during the harvest periods (Fig. 3.27B). The Cl^- levels in the winery wastewater showed two distinct peaks where the permissible maximum norm of 150 mg/L for continuous irrigation of grapevines (Howell & Myburgh, 2013 and references therein) was exceeded. One of these peaks occurred in November 2011, whereas the second coincided with the harvest period in 2013 (Fig. 3.27B).

<u>Phosphorus</u>: The variation in P could not be related to a specific activity in the winery (Fig. 3.27B). Since the levels of P in the wastewater were generally low throughout the study period, land application of the wastewater would not make a significant contribution to the P requirements of crops.

<u>pH</u>: Except during the harvest periods, the wastewater pH was most of the time within the legal requirement for wastewater irrigation as stipulated in the Department of Water Affairs (2013) General Authorization (Fig. 3.27C). Based on the foregoing, the soil was irrigated with suitable water with regards to pH, except during the harvest periods when the wastewater became acidic.

<u>COD</u>: Annually, the wastewater COD tended to peak during the harvest period (Fig. 3.27D). This confirmed that the crushing and wine making processes generated wastewater containing high levels of COD. The winery wastewater COD was considerably higher than 400 mg/L throughout the study period (Fig. 3.27). Therefore, the wastewater did not comply with the legislation for disposal through land application.

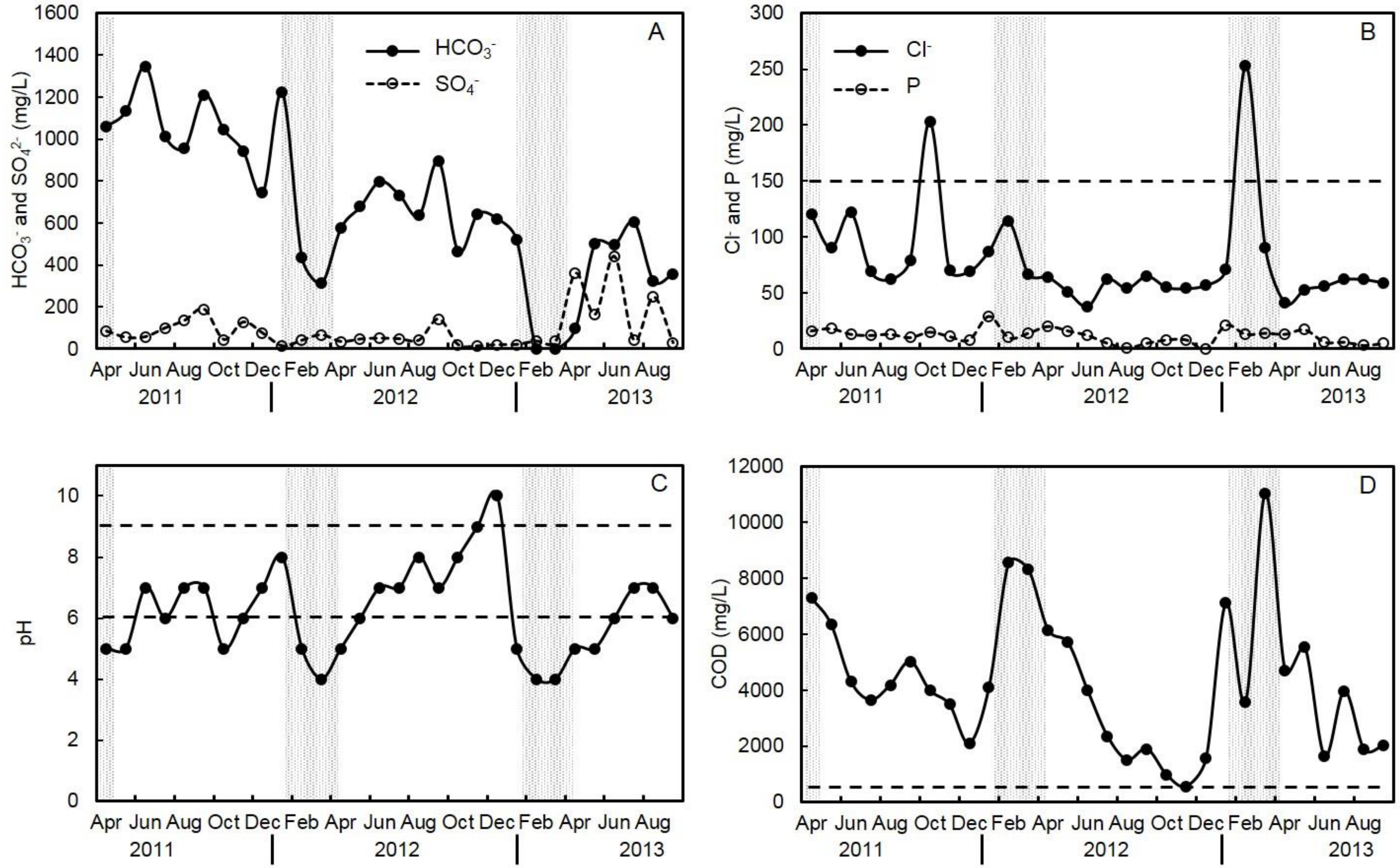

Figure 3.27. Temporal variation in (A) HCO_3^- and SO_4^{2-}, (B) Cl^- and P, (C) pH and (D) chemical oxygen demand (COD) in wastewater from the winery near Stellenbosch. Shaded columns indicate the harvest periods. Dashed lines indicate Cl^-, pH and COD thresholds.

Furthermore, the COD frequently exceeded 5000 mg/L, *i.e.* the threshold where wastewater may not be used for irrigation, or any other land application (Department of Water Affairs, 2013).

<u>Iron</u>: The Fe levels in the winery wastewater showed three distinct peaks where it exceeded the maximum acceptable water quality norm of 5 mg/L for continuous irrigation of grapevines (Howell & Myburgh, 2013 and references therein) (Fig. 3.28). It must be noted that two of these peaks coincided with the harvest periods in 2012 and 2013, respectively. At this stage, there is no explanation for the latter trend, or the source of the Fe.

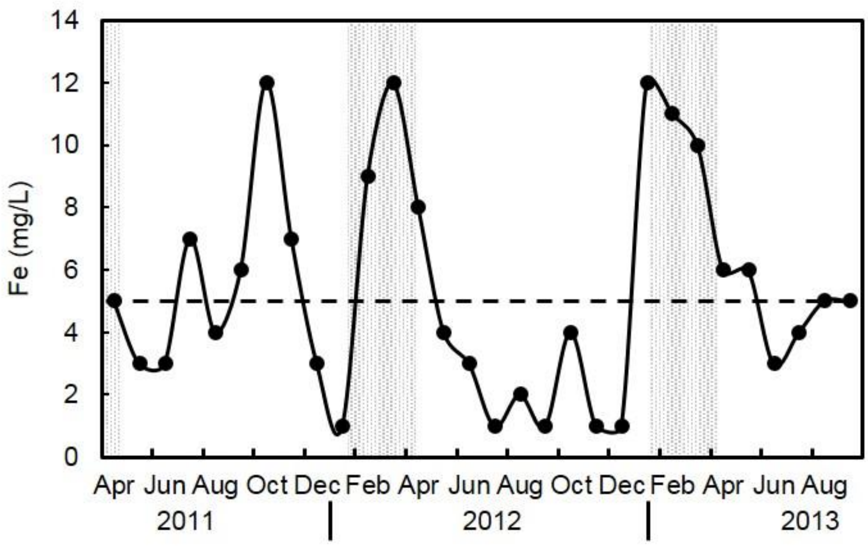

Figure 3.28. Temporal variation of iron in wastewater from the winery near Stellenbosch. Shaded columns indicate the harvest periods. Dashed line indicate the maximum Fe level for continuous irrigation.

<u>TDS</u>: The TDS variation in the winery wastewater could not be related to a specific activity in the winery, but it tended to decline during the study period (Fig. 3.29). However, at the end of the study period the TDS was slightly lower than 450 mg/L, *i.e.* the upper threshold for unrestricted use for irrigation (Ayers & Westcot, 1994).

3.3.2.2. Rainfall and volumes of wastewater applied

Mean monthly rainfall was typical for a Mediterranean climate (Fig. 3.30). Similar to Rawsonville, the July rainfall was abnormally low in all the winters. Winter rainfall, *i.e.* from April to September, amounted to 325 mm, 500 mm and 590 mm in 2011, 2012 and 2013, respectively.

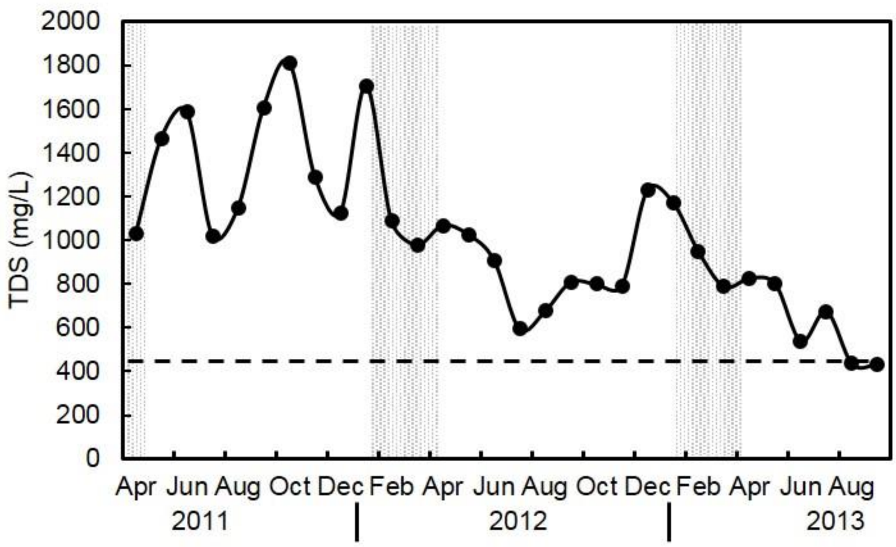

Figure 3.29. Temporal variation of TDS in wastewater from the winery near Stellenbosch. Shaded columns indicate the harvest periods. The dashed line indicates the limit for unrestricted irrigation use (Ayers & Westcot, 1994).

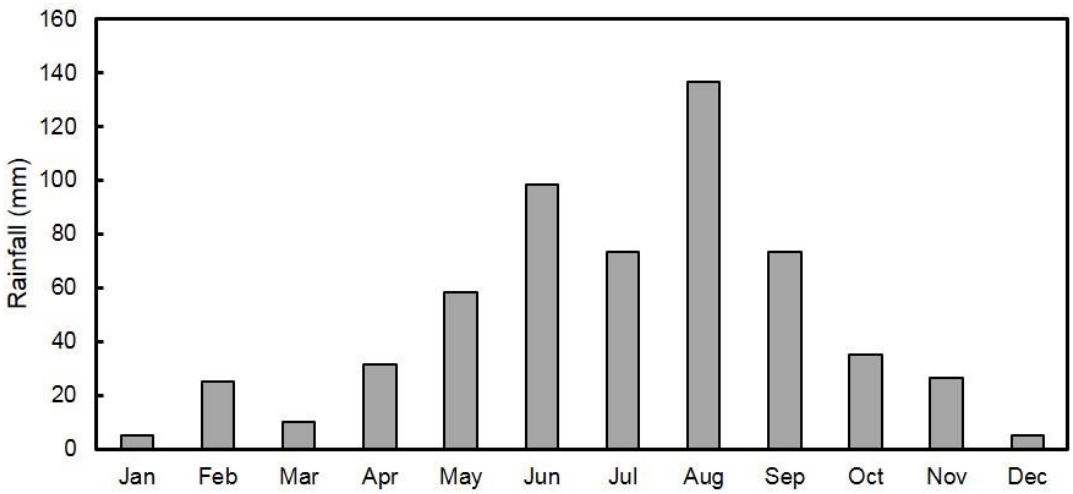

Figure 3.30. Mean monthly rainfall during the study period at a winery near Stellenbosch.

As expected, wastewater irrigations increased from December until March (Fig. 3.31). During the peak of the harvest period, in March, *c.* 30 mm irrigation was applied per day. The irrigation volumes remained relatively high in winter and began to decline from October to a minimum in December when the soil received only *c.* 1 mm wastewater per day. Total irrigation applied during winter, *i.e.* from April to September, amounted to 2670 mm, 4200 mm and 3820 mm in 2011, 2012 and 2013, respectively. Based on the foregoing, the soil received the highest irrigation plus rainfall in the winter of 2012, followed by 2013 and then 2011. Similar to Rawsonville, application of high volumes of winery wastewater caused dieback of the grass in the study plot (Fig 3.32).

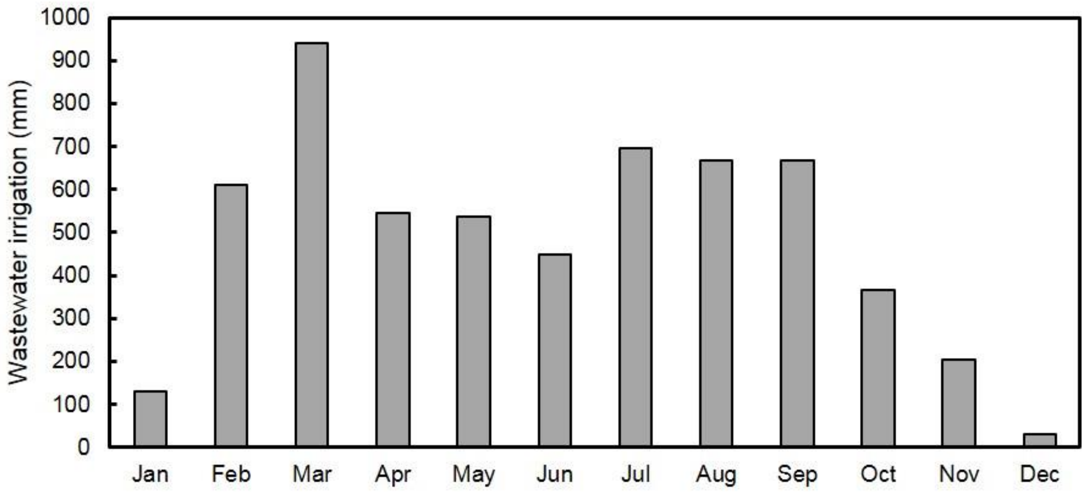

Figure 3.31. Mean monthly wastewater applied during the study period at a winery near Stellenbosch.

Figure 3.32. Disposal of volumes of winery wastewater caused dieback of the grass in the plot at a winery near Stellenbosch.

3.3.2.3. Soil chemical status

3.3.2.3.1. Initial soil chemical status

The soil status at the beginning of the study was acidic with average pH of 4.6 for the profile (Table 3.6). The Bray IIP level was acceptable throughout the soil profile, although it seemed slightly high for a sandy soil. The Na_{extr} was relatively low throughout the profile compared to K_{extr} and Ca_{extr} which seemed to dominate the exchange capacity (Table 3.6). The EPP' was relatively high compared to the ESP' which was less than 10% throughout the profile.

Table 3.6. Chemical status of the Kroonstad soil that was irrigated with winery wastewater on a new grazing paddock near Stellenbosch before the study began.

Depth (cm)	$pH_{(KCl)}$	Bray IIP (mg/kg)	Basic extractable cations ($cmol_c.kg^{-1}$)				EPP' (%)	ESP' (%)
			Na^{+}_{extr}	K^{+}_{extr}	Ca^{2+}_{extr}	Mg^{2+}_{extr}		
10	4.4	50	0.17	0.6	1.1	0.3	28.1	8.2
20	4.6	54	0.13	0.5	0.7	0.2	31.4	8.4
30	4.4	55	0.10	0.4	0.5	0.1	35.6	9.2
60	4.7	42	0.10	0.4	0.6	0.1	29.8	8.1
90	5.0	31	0.12	0.5	0.8	0.1	30.4	7.6

3.3.2.3.2. Soil chemical status during the study period:

Organic carbon: the organic C content in the 0-10 cm was substantially higher compare to the deeper layers (Fig. 3.33). During soil classification, visual observation revealed that this layer was rich in organic matter. Consequently, the 0-10 cm layer was classified as an overburden (Appendix 3.2). The initial decline of soil organic C in the 0-10 cm layer up to November 2011 was somewhat unexpected. Following the initial decline, the organic C steadily increased up to November 2013. However, the level of organic C still remained below the initial content in March 2011. This indicated that the breakdown of the overburden organic matter was more rapid than the addition of organic carbon through wastewater addition. The organic matter in the 10-20 cm layer showed a similar trend, except that the level at the end of the study period was slightly higher than the initial value (Fig. 3.33). The organic carbon in the deeper layers tended to remain constant over the two-and-a-half-year period.

Since the organic carbon in the deeper layers remained almost unchanged, it is unlikely that organic carbon could have leached from the 0-10 cm in spite of the high irrigation plus rainfall (Fig. 3.34).

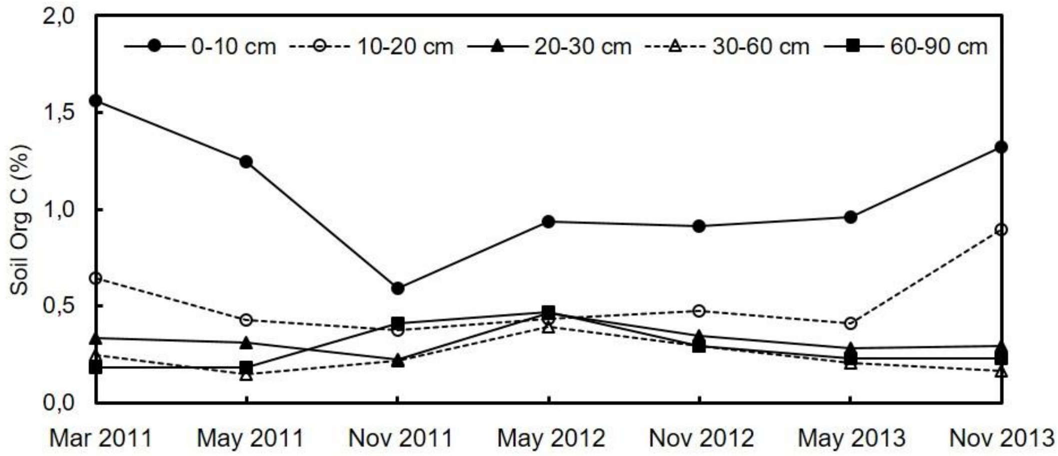

Figure 3.33. Temporal variation in soil organic C where winery wastewater was applied to a Kroonstad soil near Stellenbosch.

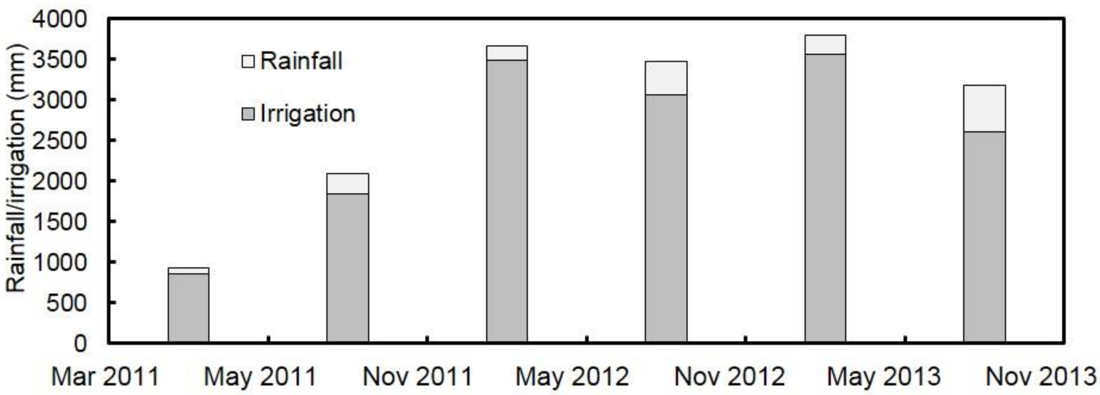

Figure 3.34. Temporal variation in rainfall plus winery wastewater irrigation as measured near Stellenbosch.

<u>**Potassium**</u>: Following an initial increase, in K_{extr} the 0-10 cm layer of this particular soil that was not previously irrigated with winery wastewater, remained relatively constant (Fig. 3.35). However, a slight increase occurred between November 2011 and May 2012. Since this trend also occurred down to a depth of 90 cm. This suggested that the high irrigation plus rainfall had leached some of the applied K^+ into the deeper layers. The K_{extr} in the 60-90

cm layer showed a steady incline over the study period which indicated that the leached K was probably steadily accumulating in the deepest layer (Fig. 3.35).

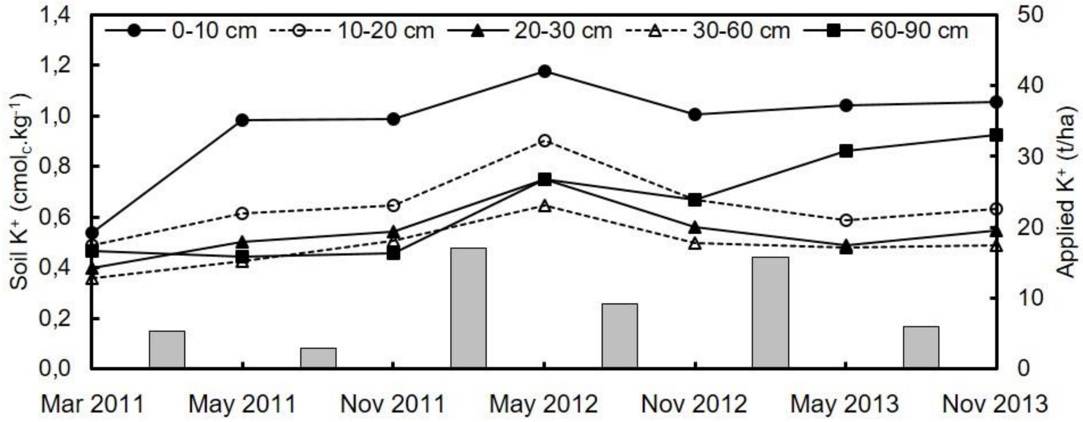

Figure 3.35. Temporal variation in soil extractable K^+ and amount of K^+ applied *via* winery wastewater to a Kroonstad soil near Stellenbosch. Vertical columns indicate amounts of applied K^+.

Since there was little change in K^+ levels with depth throughout the profile, it suggested that most of the applied K^+ was leached beyond 90 cm. in fact, seasonal soil K^+ balances showed that substantial amounts of K^+ remained in solution, and was leached (Table 3.7). Furthermore, the cumulative leached K^+ was linearly related to the cumulative irrigation plus rainfall (Fig. 3.36). Due to the low clay content of the soil (Table 3.1), the exchange complex could not retain large amounts of K^+. Therefore, leaching of K^+ beyond 90 cm was not inhibited. Leaching of K^+ in sandy or coarse textured soils during winter rainfall reduces the risk of accumulation and dispersion but it increases environmental risks such as groundwater recharge and/or lateral flow into other fresh water resources.

A previous study showed that the K^+ accumulation in soil upon winery wastewater irrigation could be high if it is not absorbed by plants, but adsorbed to soil particles thereby reducing the possibility of leaching (Arienzo *et al*., 2009b). Visual observations revealed that the roots of the grass did not extend beyond 30 cm depth. This suggested that the large amounts of the K^+ that was applied *via* the wastewater could not be utilized by the grass, since it had died back.

Table 3.7. Seasonal soil K^+ balances in the 0-90 cm depth of a sandy Kroonstad soil that was irrigated with winery wastewater near Stellenbosch.

Period	Soil K^+ (kg/ha)		Applied K^+	K^+ loss	Leached K^+
	Beginning	End	(kg/ha)	(kg/ha)	(%)
Mar 11 - May 11	2457	2633	5236	5060	97
May 11 - Nov 11	2633	2984	2883	2532	88
Nov 11 - May 12	2984	4154	17030	15860	93
May 12 - Nov 12	4154	3452	9105	9807	108
Nov 12 - May 13	3452	3686	15751	15517	99
May 13 - Nov 13	3686	3744	5934	5876	99

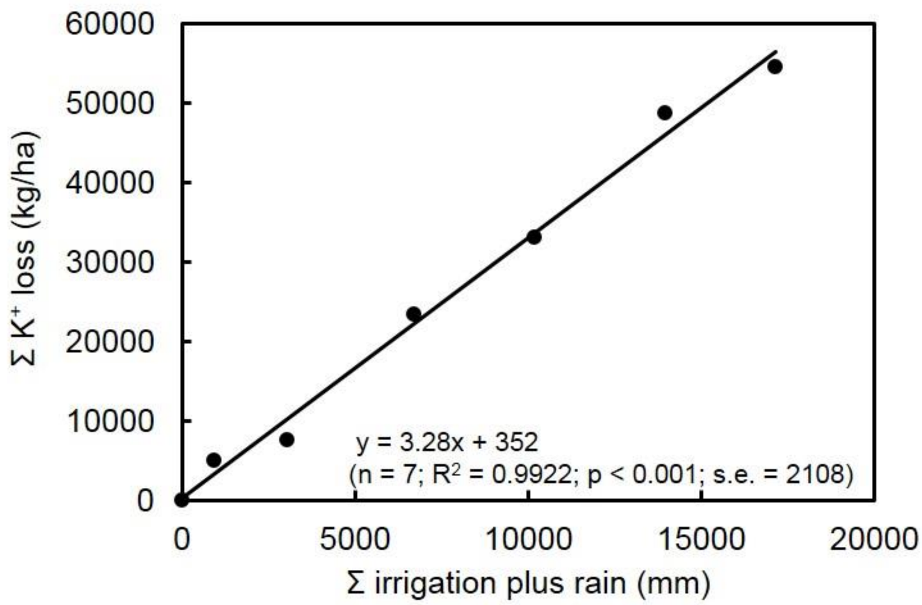

Figure 3.36. Effect of cumulative (Σ) irrigation plus rain on cumulative K^+ losses beyond 90 cm depth where a Kroonstad soil was irrigated with winery wastewater for two and a half years near Stellenbosch.

The negative effect of high levels of K^+ ions on soil structure is well documented (Levy & Torrento, 1995 and references therein). However, knowledge on the effect of high levels of K^+ in soil on soil structure stability due to winery wastewater irrigation is limited (Arienzo *et al*., 2009a).

Sodium: Except for an initial increase in May 2011, Na^{+}_{extr} tended to decline steadily throughout the study period, particularly in the 0-10 and 10-20 cm layers (Fig 3.37). The decline was probably due to the small amounts of Na^{+} being applied *via* the winery wastewater. However, the fact that the specific winery had reduced the use of Na^{+} based cleaning detergents since 2012 was probably the primary reason for decline in Na^{+}_{extr} but accumulation over time is expected. The Na_{extr} trend was in line with the levels of wastewater Na^{+} which were 41mg/l and 46.2 mg/l, respectively, during 2012 and 2013 (Tables 3.10 & 3.11).

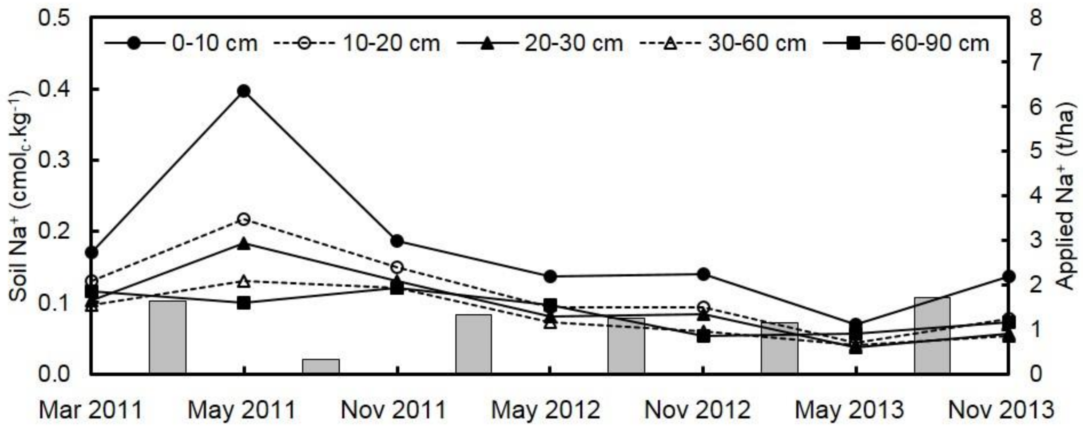

Figure 3.37. Temporal variation in soil extractable Na^{+} and amount of Na^{+} applied *via* winery wastewater to a Kroonstad soil near Stellenbosch. Vertical columns indicate amounts of applied Na^{+}.

Since there was little change in Na^{+} levels with depth throughout the profile, it suggested that most of the applied Na^{+} was leached beyond 90 cm depth. Seasonal soil Na^{+} balances confirmed that substantial amounts of Na^{+} was leached (Table 3.8). Furthermore, the cumulative leached Na^{+} was also linearly related to the cumulative irrigation plus rainfall (Fig. 3.38). Similar to K^{+}, the low clay content of the soil could not retain large amounts of Na^{+}. Therefore, leaching of Na^{+} beyond 90 cm was also not inhibited. Although, leaching of Na^{+} from sandy or coarse textured soils during winter rainfall also reduces the risk of accumulation and dispersion, it poses the same environmental risks as the large amounts of K^{+} that was leached from the soil.

Table 3.8. Seasonal balances for soil Na^+ in the 0-90 cm depth of a sandy Kroonstad soil that was irrigated with winery wastewater near Stellenbosch.

Period	Soil Na^+ (kg/ha)		Applied Na^+	Na^+ loss	Leached Na^+
	Beginning	End	(kg/ha)	(kg/ha)	(%)
Mar 11 - May 11	366	514	1645	1497	91
May 11 - Nov 11	514	411	333	436	131
Nov 11 - May 12	411	283	1331	1459	110
May 12 - Nov 12	283	221	1262	1324	105
Nov 12 - May 13	221	155	1139	1205	106
May 13 - Nov 13	155	221	1713	1647	96

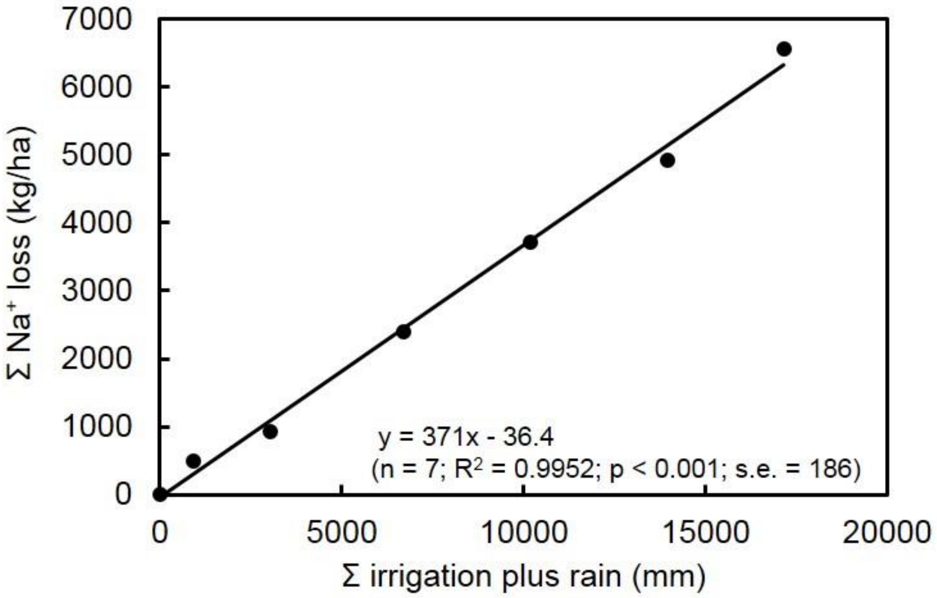

Figure 3.38. Effect of cumulative (Σ) irrigation plus rain on cumulative Na^+ losses beyond 90 cm depth where a Kroonstad soil was irrigated with winery wastewater for two and a half years near Stellenbosch.

Calcium: Although application of winery wastewater did not increase soil Ca_{extr} over the study period, this cation did show limited fluctuations (Fig 3.39). It seemed that higher applications during the harvest period reflected in the Ca_{extr}, particularly in 2012. Since the applied Ca^{2+} was substantially lower than amounts of K^+ and Na^+, it is unlikely that the Ca^{2+} would affect the EPP' or ESP' significantly. Therefore, the bivalent cation will probably not counter soil structural problems caused if the wastewater contains high levels of K^+ and Na^+ compared to Ca_{extr}. Furthermore, application of winery wastewater is unlikely to have any benefits of Ca^{2+} supply to plants because the wastewater contained only small quantities of this element.

Magnesium: The Mg^{2+}_{extr} in all layers only showed limited fluctuation over the study period (Fig. 3.40). However, this was not consistently related to the variations in the amount applied Mg^{2+}. Similar to Ca^{2+}, the low levels of Mg^{2+} were unlikely to counter the negative effects of high K^+ and Na^+ applications on EPP' or ESP', and consequently on soil physical conditions.

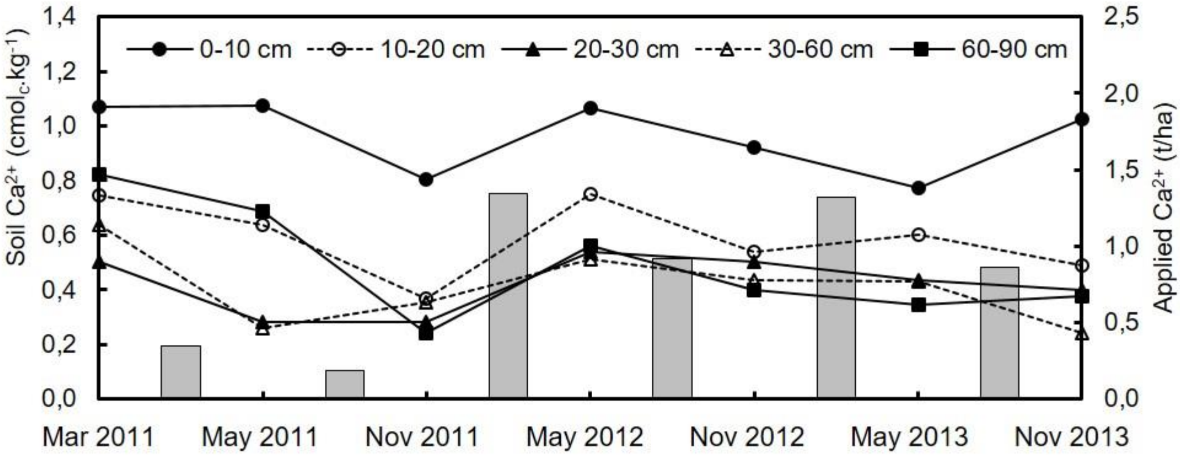

Figure 3.39. Temporal variation in soil extractable Ca^{2+} and amounts of Ca^{2+} applied *via* winery wastewater to a Kroonstad soil near Stellenbosch. Vertical columns indicate amounts of applied Ca^{2+}.

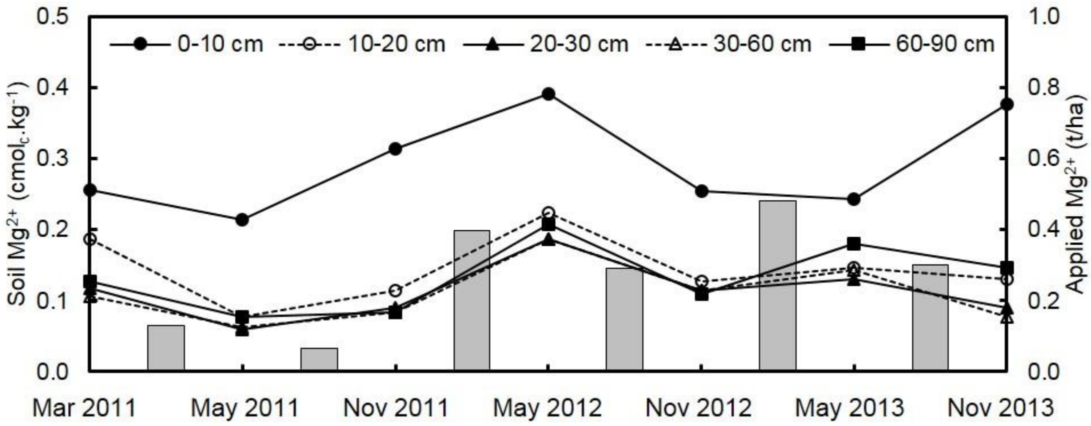

Figure 3.40. Temporal variation in extractable Mg^{2+} and amount of Mg^{2+} applied *via* winery wastewater to a Kroonstad soil near Stellenbosch. Vertical columns indicate amounts of applied Mg^{2+}.

EPP': With the exception of the 0-10 cm layer, the soil EPP' showed a steady increase over the study period (Fig. 3.41). The steepest increase occurred in the 60-90 cm layer. Since the Ca^{2+} and Mg^{2+} remained relative constant over the study period, the EPP' increase was probably due to the decline in Na^{+}_{extr} when this specific winery started to use less Na^{+} based cleaning agents.

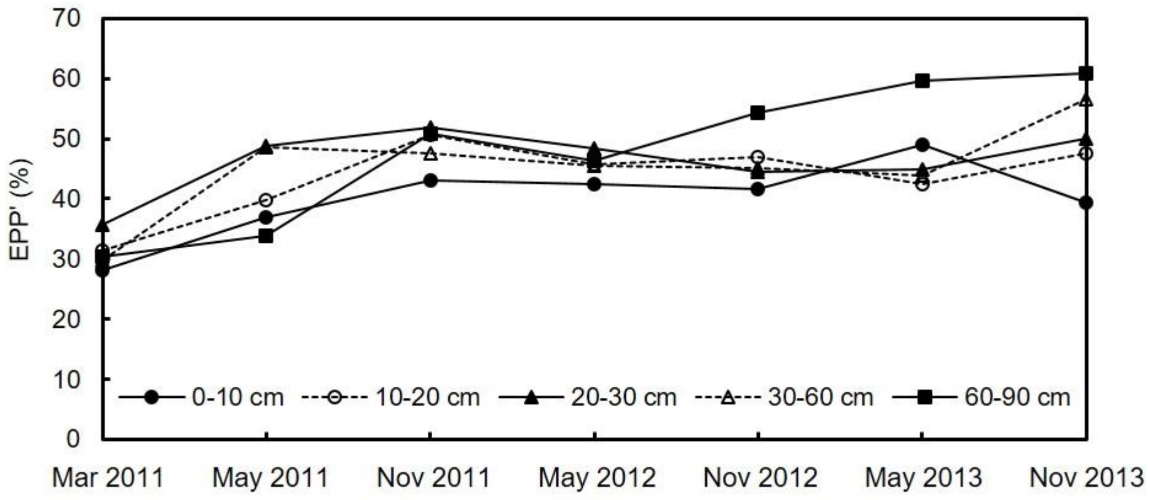

Figure 3.41. Temporal variation in the extractable potassium percentage (EPP') where winery wastewater was applied to a Kroonstad soil near Stellenbosch.

ESP': As expected, the soil ESP' followed the same trend as the Na^{+}_{extr} (Fig. 3.42). The ESP' showed an increase in May 2011, except in the 60-90 cm layer. This was followed by a steady decline until the end of the study period. Consequently, the ESP' remained below 15% in all the soil layers, except in May 2011. These results confirmed the positive effect of sound winery management practices on the reduction of the potential sodicity hazard if the water is to be used for irrigation of agricultural crops.

EC_e: The EC_e increased with soil depth throughout the study period (Fig. 3.43). The salt content in the 0-10 cm layer tended to remain almost constant over the study period, whereas it tended to decrease up to November 2012 in the deeper layers. This was followed by an increase in May 2013. However, the EC_e was lower at the end of the study period compared to the initial values in all layers (Fig. 3.43). These results indicated that the salinity hazard was reduced where winery wastewater was applied under the prevailing conditions.

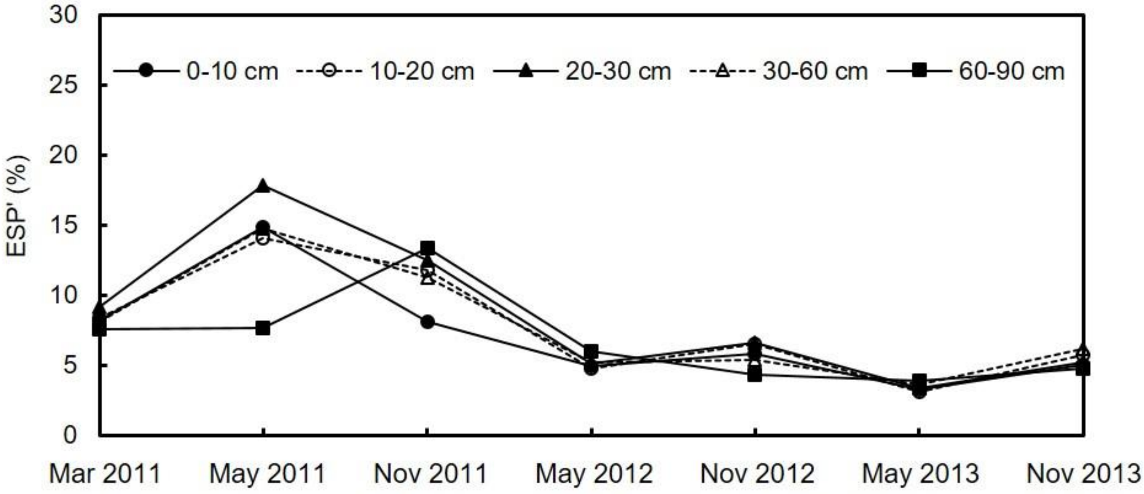

Figure 3.42. Temporal variation in the extractable sodium percentage (ESP') where winery wastewater was applied to a Kroonstad soil near Stellenbosch.

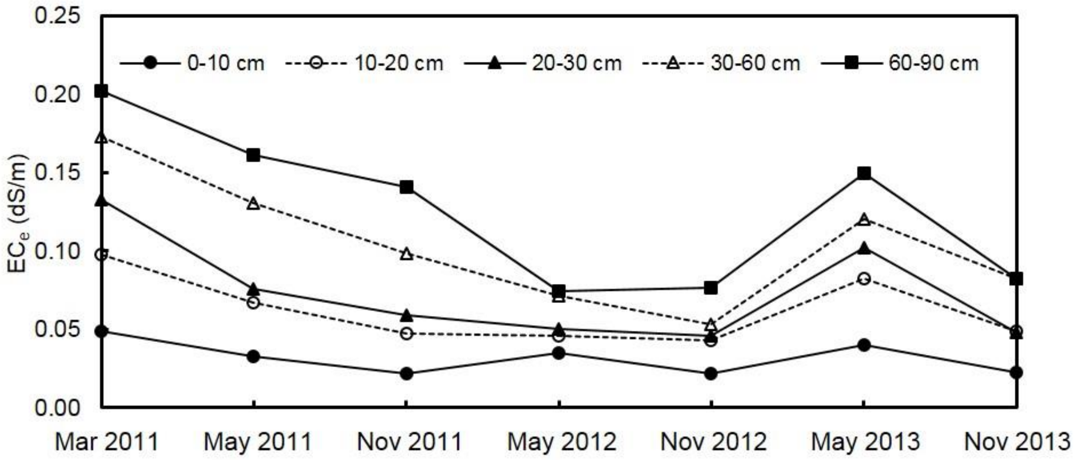

Figure 3.43. Temporal variation in the electrical conductivity of the saturated soil paste (EC_e) where winery wastewater was applied to a Kroonstad soil near Stellenbosch.

pH$_{(KCl)}$: The soil was acidic, *i.e.* the pH$_{(KCl)}$ was below 5.5, in all the layers throughout the study period (Fig. 3.44). Furthermore, the pH$_{(KCl)}$ tended to increase during the harvest period, followed by a decline in winter. This trend was notably less prominent in the 60-90 cm layer. However, the overall effect of irrigation with winery wastewater was that the pH$_{(KCl)}$ increased in all layers over the study period (Fig. 3.44). Application of winery wastewater increased the soil pH$_{(KCl)}$ from 4.6 to 5.0 in the topsoil and from 5.0 to 5.3 in the subsoil. The pH$_{(KCl)}$ increase in the topsoil means that organic materials supplied by winery wastewater could be the source of the pH$_{(KCl)}$ increase. In contrast, leaching of salts into the deeper soil layers could have increased pH$_{(KCl)}$.

According to Rukshana *et al*. (2012), soil pH increased when organic anions were mineralised and H^+ ions were consumed following winery wastewater application. Although application of winery wastewater increased soil pH by more than 0.2 units, the soil remained acidic under the prevailing conditions.

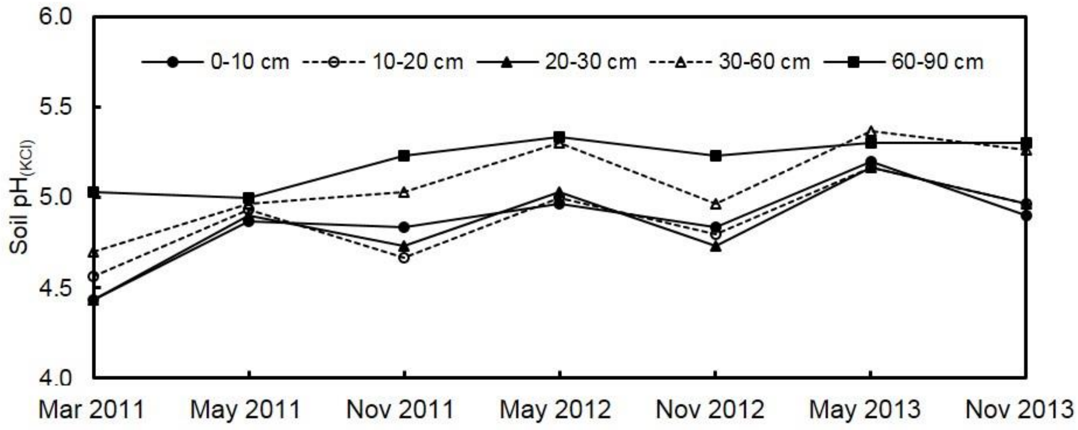

Figure 3.44. Temporal variation in soil $pH_{(KCl)}$ where winery wastewater was applied to a Kroonstad soil near Stellenbosch.

<u>Phosphorus</u>: The soil P did not show any trends that could be related to seasonal variation in the volumes of winery wastewater applied, or the level of P in the water (Fig. 3.45). Although limited fluctuations occurred, the soil P tended to increase slightly over time, except in the 60-90 cm layer. The latter suggested that P did not leach into the subsoil under the prevailing conditions. The soil P content was well above the minimum requirement for vineyards (Conradie, 1994).

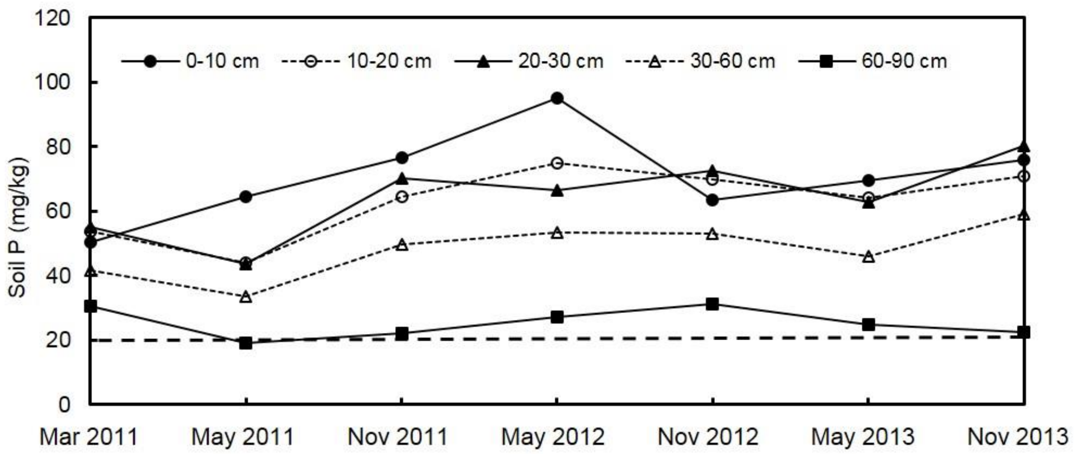

Figure 3.45. Temporal variation in soil P where winery wastewater was applied to a Kroonstad soil near Stellenbosch. Dashed line indicate the proposed P norm for grapevines (Conradie, 1994).

3.4. CONCLUSIONS AND RECOMMENDATIONS

It is important to note that the study represented the worst case scenario, *i.e.* the winery wastewater disposal was not carried out in a bigger paddock. Consequently, high volumes of wastewater irrigation were applied on a single plot, particularly in the harvest period and winter. Visual observations revealed that this caused waterlogging in the subsoil of a Longlands soil near Rawsonville, as well as the Kroonstad soil near Stellenbosch. Water movement in the Longlands would probably be vertical, whereas would be lateral above the G horizon for the Kroonstad. Although the winery wastewater contained high loads of organic C, it did not accumulate in any of the two soils. This suggested that aerobic conditions in the topsoil allowed decomposition of the applied organic matter during December when the wastewater irrigation volumes were at lowest and temperatures were high.

Due to the high volumes of wastewater irrigation plus rainfall, the inevitable over-irrigation leached large amounts of cations, particular K^+ and Na^+, beyond 90 cm depth in the Longlands and Kroonstad soils. This was confirmed by the fact that the EC_e remained fairly constant during the study period. Unfortunately, the leached elements are bound to end up in natural water resources in the long run. Irrigation with the winery wastewater did not have a pronounced effect on soil $pH_{(KCl)}$, except for a slight increase in the Kroonstad soil near Stellenbosch. This was probably due to the decomposition of organic matter, and the fact that the applied salts were leached beyond 90 cm depth. The study confirmed that injudicious irrigation with untreated winery wastewater poses a serious environmental hazard, particularly where crops in sandy soils are irrigated.

Due to the risks involved as discussed above, disposal of winery wastewater by means of irrigation is definitely not the ultimate solution to the problem. Land disposal can only be recommended where the wastewater application does not exceed the water requirement of the grazing crop, or any other agricultural crop. Wastewater application according to the potassium requirement of the crop is also very crucial. This means that the wastewater needs to be distributed on an area of land that is big enough so that the daily applications does not cause over-irrigation. Therefore, sound wastewater management can only be achieved by means of irrigation scheduling based on frequent soil water content measurements. Care should be taken that the irrigation water does not leach beyond the root zone. The soil chemical status should be determined at least annually. The basis to which wastewater should be applied for a given crop should be based on water and nutrients requirement such as potassium. Depending on the type of soil and quality of wastewater, each winery will determine the size of land needed for irrigation with wastewater high on potassium. The winery will also have to consider the electricity costs if wastewater needs to be pumped from nearby farms in order to be utilized for a crop requirement.

Based on the foregoing, it is essential that future research should focus on selecting halophytic crops that are capable of absorbing the applied elements, particularly K^+ and Na^+, if land disposal of winery wastewater is the only option. Preferably, the foliage and roots or tubers should be removed from the land when the crop is harvested. The effects of K:Na ratio in diluted or undiluted winery wastewater on soil structure stability, potassium availability and leaching of elements also needs to be addressed by continued research. Since the climate, particularly rainfall, will affect the accumulation and/or leaching of the elements, it is important that the research is carried out in field studies.

CHAPTER 4: DESIGN OF A POT EXPERIMENT TO STUDY THE EFFECT OF IRRIGATION WITH DILUTED WINERY WASTEWATER ON FOUR DIFFERENTLY TEXTURED SOILS

4.1 INTRODUCTION

The negative effects of irrigation with winery wastewater on soils are well documented (Bond 1998; Papini, 2000; Mulidzi, 2001; Arienzo *et al*., 2009a; Christen *et al.*, 2010; Laurenson & Houlbrooke, 2011; Laurenson *et al*., 2012; Mosse *et al.*, 2011; Arienzo *et al*, 2012). To comply with intensified environmental legislation (Department of Water Affairs, 2013), the wine industry must find solutions for treatment or re-use of winery wastewater (Van Schoor, 2001). Since negative impacts on soils might be less if the winery wastewater is diluted before being re-used for irrigation, such a practice could be more sustainable compared to undiluted wastewater. However, knowledge regarding effects of diluted winery wastewater on different soils in South African grape growing regions is limited.

Determining effects of irrigation with winery wastewater on soils and crops in field experiments, requires an elaborate infrastructure, particularly if the wastewater has to be diluted to a predetermined level (Myburgh *et al.*, 2014). Field experiments are usually carried out with one specific soil type. Since different soils respond differently to winery wastewater irrigation (Mulidzi, 2001), it is essential to determine the effects of diluted winery wastewater on soils that differ pedogenically. However, it would be expensive to erect the required infrastructure for a range of soils. A further disadvantage of field experiments is that wineries produce the bulk of their wastewater during the harvest period, *i.e.* from February to April. Therefore, field experiments can only be carried out annually during harvest. Based on the foregoing, pot experiments seem to be an alternative, since it could include a range of different soils. A further advantage is that winery wastewater can be stored in tanks which will allow experiments to be continued throughout the year if the pots are sheltered from rain. This will reduce the duration of experiments compared to ones carried out in the open field. If pot experiments are carried out correctly, drainage and subsequent leaching of elements can be avoided. The latter can be problematic and difficult to quantify under field conditions.

The objective of the study was to design and evaluate a pot experiment to determine the effects of irrigation with diluted winery wastewater on different soils.

4.2. MATERIALS AND METHODS

4.2.1 Experiment layout

Four different soils from grape growing regions in the Western Cape Province were included in the study. A sandy, alluvial soil was collected in a vineyard near Rawsonville in the Breede River valley. This soil belongs to the Longlands (Arenosol) form. A sandy, aeolic soil which belongs to the Garies form (Eutric Petric Durisol) was collected near Lutzville in the Lower Olifants River valley. A shale derived soil was collected on the Nietvoorbij Experiment farm of the Agricultural Research Council near Stellenbosch. A granite derived soil was also collected at Nietvoorbij. These soils belong to the Oakleaf (Chromic Acrisol) and Cartref (Albic, leptic Acrisol) forms, respectively. For the purpose of the study, the soils will be referred to as Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite, respectively.

The alluvial sand was collected in a vineyard, whereas the others were from uncultivated land. Composite soil samples were collected from the 0-300 mm layer at each locality and placed in plastic bags for transport and storage. The shale and granite soils were passed through a 6 mm mesh sieve to remove large fragments. Triplicate samples were collected from the composited soils for determining particle size distribution at a commercial laboratory (Bemlab, Strand). Five soil particle size classes were determined using the hydrometer method (Van der Watt, 1966). Soil textural classes were assigned according to standard diagrams of the Soil Classification Working Group (1991).

The pot experiment was carried out under a 20 m x 40 m translucent fiberglass rain shelter at ARC Infruitec-Nietvoorbij near Stellenbosch. Due to logistic constraints, irrigation water for the control treatments, as well as for wastewater dilution, could not be obtained from Lutzville and Robertson. Therefore, the control soils were irrigated with water supplied by the Stellenbosch municipality. For the wastewater treatments, winery wastewater was diluted to a chemical oxygen demand (COD) level of 3000 mg/L. The undiluted wastewater was collected from the wastewater pit at a winery near Rawsonville, and stored in a 2500 L plastic stock tank next to the rain shelter. A 500 L plastic tank was filled with municipal water. The COD in the undiluted wastewater and municipal water was measured using a spectrophotometer (Aqualitic

COD-reactor®, Dortmund) with appropriate test kits (COD, CSB, 0-15000 mg/L). The COD levels were used to calculate the volumes of winery wastewater and municipal water required to obtain the target COD level. The volume (m^3) of wastewater required from the stock tank (V_W) to obtain a certain target COD concentration (COD_T) was calculated as follows:

$$V_W = (COD_T - COD_M) \times V_T/(COD_S - COD_m) \quad \text{(Eq. 4.1)}$$

where COD_m and COD_s are the COD concentrations (mg/L) in the municipal water and the stock tank, respectively, and V_T is the tank volume (m^3). The volume of wastewater required (V_W) was pumped from the stock tank into another 500 L plastic tank where it was mixed with municipal water. The COD in the diluted wastewater was measured while the irrigations were applied.

Treatments were applied over four simulated irrigation seasons. Each season consisted of six irrigations, which was estimated as the number of irrigations a vineyard would require during the harvest period, *i.e.* when the highest volumes of wastewater are produced. Hence, a total of 24 irrigations were applied over the four simulated irrigation seasons. Each soil/water treatment combination was replicated three times in a complete randomised block design. Following each simulated season, *i.e.* after 6, 12, 18 and 24 irrigations, the soil chemical status was determined to compare the effect of irrigation with diluted winery wastewater to that of municipal water. Since soil sampling was destructive, a replication "plot" of each soil/water treatment combination consisted of four pots (Fig.4.1). At the end of each season, one of the four pots was removed for sampling.

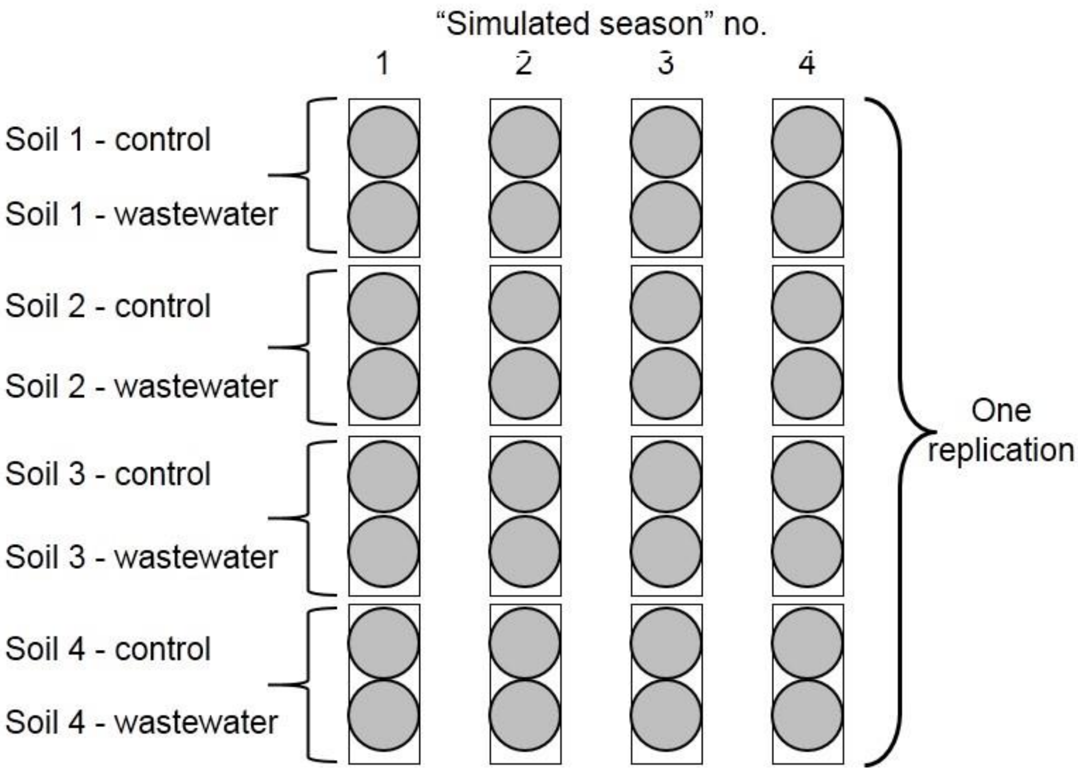

Figure 4.1. Schematic illustration of the layout of one replication of the treatment pots. In the actual experiment layout, the pots of different treatments were randomised in each replication.

4.2.2. Packing of soils to a predetermined bulk density

The day before the pots were filled, the bulked soils were moistened using municipal water to enhance compaction. Following this, the soils were mixed thoroughly and covered using plastic sheets to minimize water loss. Triplicate soil samples were collected in metal cans to determine the gravimetric water content of the bulked soils. The moist soil samples were weighed and dried in an oven at 105°C for 16 hours. Samples were removed from the oven and allowed to cool in a desiccator before they were weighed to obtain their oven-dry mass. Gravimetric soil water content (Θ_m) was calculated as follows:

$$\Theta_m = (M_w - M_d) \div (M_d - M_c) \qquad \text{(Eq. 4.2)}$$

where M_w is the mass of the moist soil plus the can, M_d is the oven-dry mass of the soil plus the can and M_c is the metal can mass. Mass percentage soil water content (SWC_m) was calculated as follows:

$$SWC_m = \Theta_m \times 100 \qquad \text{(Eq. 4.3)}$$

All the soils were packed to a bulk density (ρ_b) of 1400 kg/m^3. The mass of moist soil required to obtain this target ρ_b was calculated as follows:

$$M_p = (\rho_b \times V_s) \times (1 + SWC_m \div 100) \quad \text{(Eq. 4.4)}$$

where M_p is the mass of the moist soil that needs to be packed into the pot (g), ρ_b is the target bulk density (kg/m^3), V_s is the soil volume in cm^3. Soils were packed into 3.54 dm^3 PVC pots which consisted of 200 mm lengths of 150 mm ø PVC pipe with a wall thickness of 4 mm. The base of each pot was machined from 3 mm thick PVC sheet, and glued onto one of the open ends. A 10 mm ø hole was drilled in the bottom of each pot to allow drainage. A piece of 1.5 mm plastic fly-mesh was placed on the bottom of each pot to prevent the soil from being lost through the drainage hole. All pots were cleaned and weighed before being filled with the soil. A custom built mechanical press was used to compact the soils to the required ρ_b. The packed soil columns were only 190 mm high, *i.e.* leaving 10 mm below the upper edge of a pot free. The surface under the rain shelter was first leveled with a gravel layer to promote even distribution of the irrigation water. A layer of coarse building sand was placed on the gravel. The pots were placed onto 240 mm ø plastic saucers. The area occupied by the pot experiment was 3.7 m x 7.8 m. For each soil, four additional pots were packed. The soil in these pots were saturated using municipal water. After free drainage had stopped, the pots were weighed to obtain the mean mass for each soil at field capacity. The dry soil mass (M_{od}) in each pot (g) was calculated as follows:

$$M_{od} = V_s \times \rho_b \quad \text{(Eq. 4.5)}$$

where V_s is the volume (dm^3) of soil and ρ_b is the target bulk density (kg/m^3).

4.2.3. Irrigation system

Two 0.74 kW, 3 m^3/h pumps (Foras®, Berg River Irrigation, Paarl) were used to apply the municipal and diluted winery wastewater to the respective soil/water treatment combinations. The municipal and diluted winery wastewater passed through two 130-micron ring filters (Arkal®, Netafim, Kraaifontein) installed downstream of each pump. An eight-way manifold with a ball valve at each outlet allowed individual irrigation of the eight soil/water treatment combinations. Water was distributed through a network consisting of 17 mm ø laterals, and applied to each pot by means of a 2 L/h pressure compensating button dripper with a four-way manifold attached to it (Netafim, Kraaifontein). Four 700 mm long, 3 mm inner ø micro-tubes were attached to each dripper manifold (Fig. 4.2). In order to obtain equal flow through the four micro-tubes, an inline labyrinth-type dripper (Arrow®, Netafim, Kraaifontein) was inserted in the open end of each micro-tube to create some back pressure. To distribute the irrigation water uniformly over the soil surface, the four micro-tubes were supported by a brace placed onto each pot (Fig. 4.3). The brace, in the form of a cross, was made from two 1.8 mm x 7 mm x 205 mm galvanised metal strips. On each side of the metal strips, the last 20 mm was bent at a rectangle so that the brace fitted firmly onto a pot. The four micro-tubes were pushed through 5 mm ø holes drilled in the brace. The flow rate through each of the four micro-tubes was 0.5 L/h. The total flow rate through the four micro-tubes (Q_{drip}) was 34 mL/min.

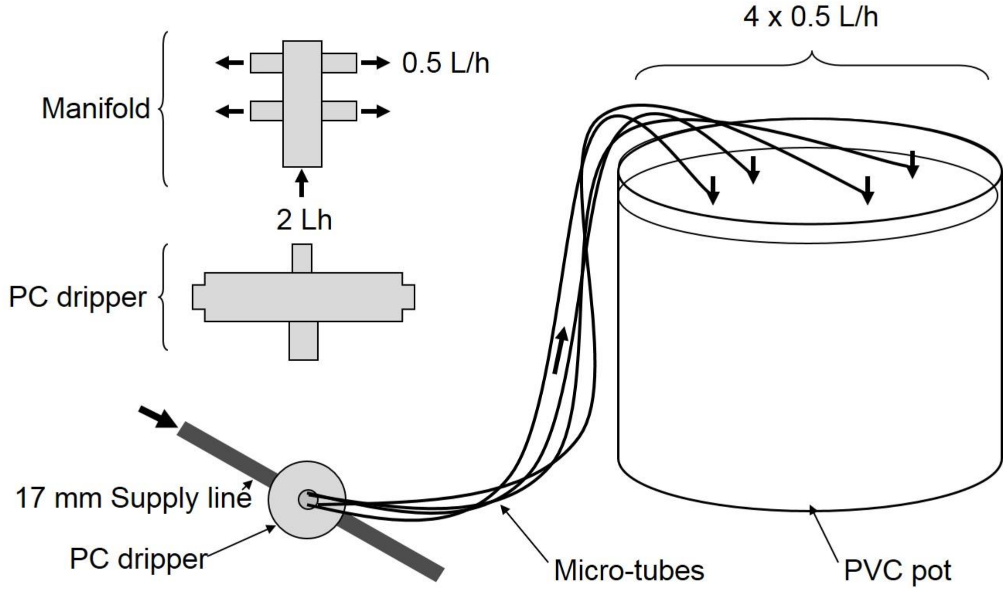

Figure 4.2. Schematic illustration pressure compensating (PC) dripper, manifold and micro-tubes to distribute water evenly in the pots.

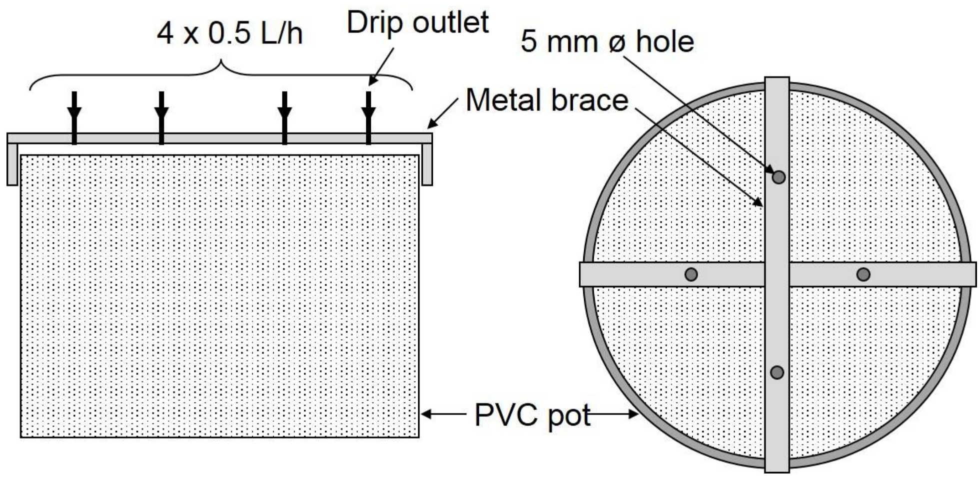

Figure 4.3. Schematic illustration of PVC pot with galvanized metal brace bearing the four micro-tubes.

4.2.4. Irrigation volumes

The volume of water applied to each soil was recorded using water meters. The mass of water in each soil at field capacity (WM_{fc}) was calculated as follows:

$$WM_{fc} = M_{fc} - M_{od} \quad \text{(Eq. 4.6)}$$

where M_{fc} is the mean pot plus soil mass at field capacity (g) and M_{od} is the pot plus oven-dry soil mass (g). To irrigate when WM_{fc} had evaporated to a specific depletion percentage (P), the pot plus soil mass at that depletion percentage (M_{depl}) was calculated as follows:

$$M_{depl} = M_{od} + (WM_{fc} \times P \div 100) \quad \text{(Eq. 4.7)}$$

Since weighing 96 pots was too laborious, only 4 representative pots per soil/water treatment, *i.e.* 32 in total, were weighed. Before weighing these pots, the braces bearing the micro-tubes were removed. An electronic balance was used to weigh the pots every second day until the mass reached M_{depl}. Assuming that the water density is 1 g/cm^3, the irrigation volume required per pot was calculated as follows:

$$V_{irr} = WM_{fc} - M_{act} \quad \text{(Eq. 4.8)}$$

where V_{irr} is the volume of water required per pot (mℓ) and M_{act} is the actual pot plus soil mass before irrigation (g). The time required to apply V_p was calculated as follows:

$$t = V_{irr} \div Q_{drip} \quad \text{(Eq. 4.9)}$$

where t is the time (min) and Q_{drip} is the total flow rate through the four micro-tubes (mL/min). The soils were irrigated when their P reached *c.* 85%. This high level of depletion was to ensure adequate soil aeration between irrigations. When pots were removed for soil chemical analyses, their irrigation water was collected in 500 mL glass beakers and discarded. This was to maintain the same irrigation system flow rate throughout the experiment.

4.3. RESULTS AND DISCUSSION

Since only topsoil was used in the study, characteristics of the deeper horizons were considered to be irrelevant. With the exception of the Stellenbosch granite soil, which had a high coarse sand fraction, fine sand dominated the sand fraction (Table 4.1). According to Van Huyssteen (1989), this particular soil contains *c.* 47% gravel, *i.e.* 2-6 mm ø, in its natural state. All soils compacted with relative ease to a ρ_b of 1400 kg/m^3. When the soils were packed into the pots, mean water contents were 14.9%, 11.7%, 12.1% and 14.5%, respectively, for the Rawsonville sand, Lutzville sand, Stellenbosch shale soil and Stellenbosch granite soil. Irrigation amounts applied to the Rawsonville sand, Lutzville sand, and Stellenbosch shale soil over the four simulated seasons were comparable, but the Stellenbosch granitic soil received substantially less water (Table 4.2). As expected, the COD in the municipal water was substantially lower compared to the diluted winery wastewater (Table 4.3). The COD in the diluted winery wastewater was comparable between the four simulated seasons, and was reasonably close to the target level of 3000 mg/L.

Table 4.1. Locality, soil form, particle size distribution (≤ 2 mm) and textural class for the four soils included in the study.

	Rawsonville sand	Lutzville sand	Stellenbosch shale	Stellenbosch granite
Latitude	-33.693698°	-31.558906°	-33.911717°	-33.917296°
Longitude	19.322569°	18.353115°	18.871152°	18.864484°
Clay (<0.002 mm)	3.3	0.4	20	13
Silt (0.002-0.02 mm)	1	1	13	17
Fine sand (0.02-0.2 mm)	60	69	50	33
Medium sand (0.2-0.5 mm)	29	26	5	3
Coarse sand (0.5-2 mm)	8	2	12	35
Soil textural class	Fine sand	Fine sand	Fine sandy clay loam	Coarse sandy loam

Table 4.2. Total irrigation amounts applied to four different soils during four simulated seasons.

Soil	Irrigation applied (mm/season)				Total
	Season 1	Season 2	Season 3	Season 4	
Rawsonville sand	291	289	287	289	1156
Lutzville sand	281	282	282	281	1126
Stellenbosch shale	246	250	246	245	987
Stellenbosch granite	181	180	184	183	728

Table 4.3. Variation in chemical oxygen demand (mg/L) in the water used for the pot experiment.

Water source	Season				Mean
	1	2	3	4	
Municipal water	35	25	26	26	28±4
Diluted winery wastewater	3149	3257	3243	3190	3210±43

The soil water contents at field capacity of the soils were comparable, except for the Stellenbosch granite soil (Fig. 4.4). This indicated that this particular soil had a lower water holding capacity compared to the other soils. The lower water holding capacity of the Stellenbosch granite soil was probably due to the high coarse sand content (Table 4.1). Initially, the soil water content was restored to field capacity following irrigation in all soils. However, in the case of the Stellenbosch granite, field capacity was only restored following the first two irrigations (Fig. 4.4D). From the third irrigation onwards, visual observation revealed that the irrigation water ponded on the soil surface due to poor water infiltration. Consequently, the target soil water depletion level was reached following irrigation, but field capacity could not be restored (Fig. 4.4D). Although actual soil water content was not measured in the pots, it can be assumed that only the upper section of the profile in the Stellenbosch granite soil was wetted.

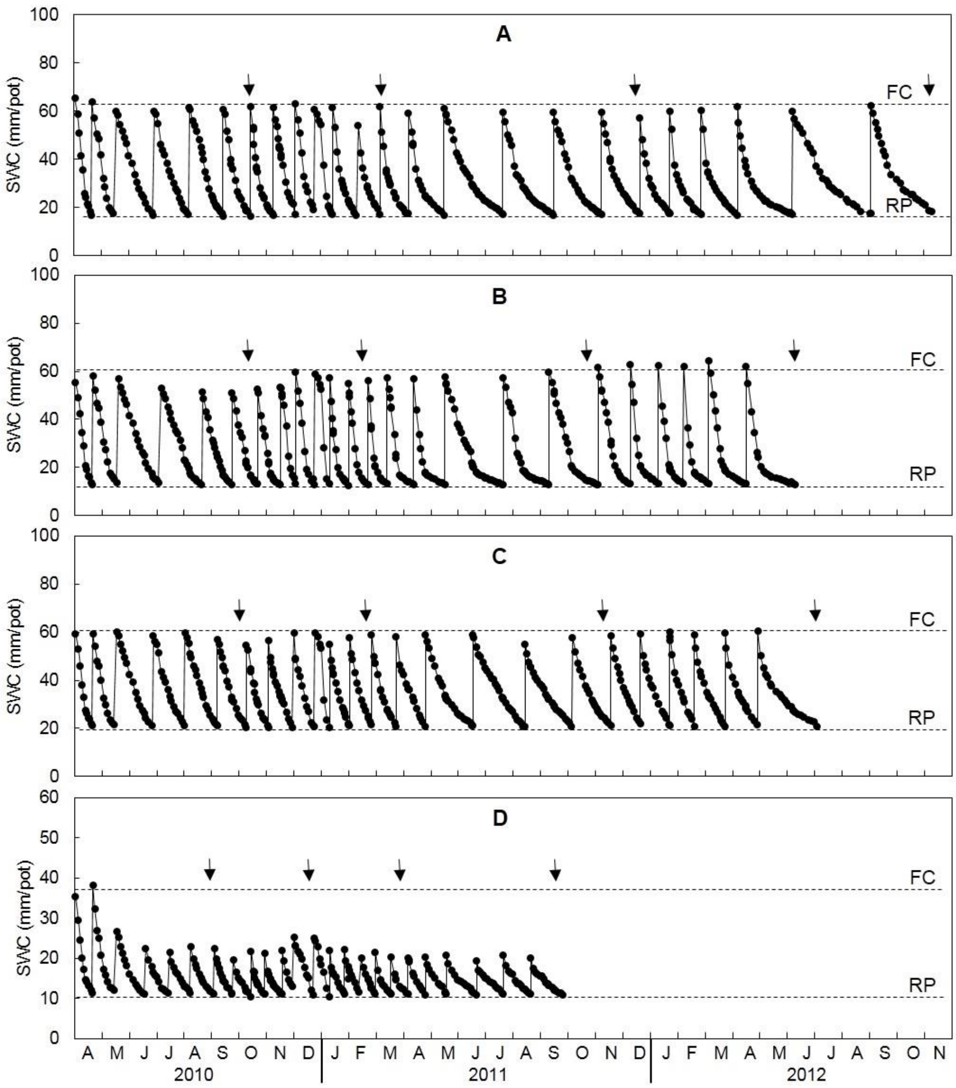

Figure 4.4. Temporal variation in soil water content (SWC) in (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils measured in a pot experiment. Arrows indicate when soil chemical status was determined after each of the simulated seasons. "FC" and "RP" indicate field capacity and refill point, respectively.

Although the level of COD differed substantially between the municipal and winery wastewater (Table 4.3), water infiltration problems occurred where municipal, as well as winery wastewater were applied to the granite soil. The sodium adsorption ratios in the municipal and winery wastewater were 0.8±0.1 and 4.6±0.6, respectively (unpublished data). This confirmed that poor water quality did not cause the problem. Since the soil was not saline, irrigation with low salinity water could not have caused the problem in the case of the clean water treatments. When irrigated with clean river water and a range of diluted winery wastewaters, the near-saturation hydraulic conductivity of this particular soil was considerable lower compared to the other soils, irrespective of the level of water quality (Howell & Myburgh, 2014).

With the exception of the Stellenbosch granite soil, the soil water content was managed between field capacity and the refill point (Fig. 4.4). This indicated that the soils were well-aerated between irrigations. Since the lower part of the Stellenbosch granite must have remained dry, it implied that this soil was also well-aerated between irrigations. Visual observation revealed that no drainage occurred after irrigations had been applied. Therefore, it can be assumed that no leaching occurred of elements applied *via* the municipal or diluted winery wastewater. The foregoing indicated that the lysimetric approach provided an accurate measure of the irrigation volumes required. It is important to note that the pot experiment was completed in approximately two and a half years, whereas it would have taken four years to do the wastewater irrigations in a field experiment. Effects of irrigation with diluted winery wastewater on the chemical status of the four soils will be presented in subsequent chapters.

4.4. CONCLUSIONS

It was possible to subject more than one soil to irrigation with diluted winery wastewater by using a single mix and irrigation infrastructure. Since the pot experiment could be continued under the rain shelter during winter, results were obtained quicker compared to an open field study. Although only representative pots were weighed, the procedure was still time consuming. Therefore, it is recommended that load cells are used to record daily mass losses automatically. Automatic recording will also be useful for determining mass losses if experiments are carried out with potted plants.

CHAPTER 5. EFFECT OF IRRIGATION WITH DILUTED WINERY WASTEWATER ON CATIONS, PH AND PHOSPHORUS IN FOUR DIFFERENTLY TEXTURED SOILS

5.1. INTRODUCTION

Changes in environmental legislation (Department of Water Affairs, 2013) put pressure on the wine industry to find solutions for treatment or use of winery wastewater (Van Schoor, 2001a). This initiated the development of guidelines for the management of wastewater and solid waste at wineries (Van Schoor, 2005). In many cases, shortage of good quality water leads to an increasing need to irrigate with poor quality water such as saline groundwater, drainage water and treated wastewater (Jalali *et al.*, 2008). The impact of using untreated industrial and municipal wastewater for agricultural irrigation is well-documented (Bond 1998; Papini, 2000; Mulidzi, 2001; Arienzo *et al.*, 2009b; Christen *et al.*, 2010; Laurenson & Houlbrooke, 2011; Mosse *et al.*, 2011; Arienzo *et al.*, 2012; Laurenson *et al.*, 2012; Howell & Myburgh, 2014; Walker & Lin, 2008).

Disposal of winery wastewater through land application has been practiced for many years (Mulidzi, 2001; Laurenson & Houlbrooke, 2011). Effective disposal of wastewater depends on the irrigation technology, as well as on soil properties (Oron *et al.*, 1999). An earlier study confirmed that the impacts of using undiluted winery wastewater for irrigation differ substantially between soil types (Mulidzi, 2001). Under some circumstances irrigation with winery wastewater can have a beneficial effect. It was also suggested that using K^+-rich wastewater could enhance soil fertility (Mosse *et al.*, 2011). In Australia, continued irrigation of pastures with winery wastewater resulted in an accumulation of K^+ to levels that leached into the groundwater and other water resources (Christen *et al.*, 2010). In addition, it was observed that using winery wastewater for irrigation of poorly drained soils can lead to salinisation and water-logging, reducing the long-term sustainability of the land for agriculture (Christen *et al.*, 2010).

Replacement of bivalent Ca^{2+} and Mg^{2+} by monovalent K^+ and Na^+ during continuous irrigation can potentially lead to the breakdown of the soil structure. Exchangeable Na^+ in soils tends to increase where wastewaters containing high levels of Na^+ are used for irrigation (Lieffering & McLay, 1996). Where wineries use Na^+ based cleaning agents, *e.g.* sodium hydroxide (NaOH), accumulation of monovalent cations, such as Na^+, on the exchange sites has the potential to degrade soil structure through clay dispersion and flocculation (Mosse *et al.*, 2011). Consequently, soil hydraulic conductivity can be reduced where winery wastewater is used for irrigation (Laurenson *et al*., 2012). Indications of poor aeration and water infiltration observed in various soils where winery wastewater was used for irrigation, were attributed to structural degradation caused by high Na^+ concentrations added to the soil (Mulidzi *et al*., 2009b). This was confirmed when irrigation with diluted winery wastewater reduced the hydraulic conductivity of differently textured soils (Howell & Myburgh, 2014). Using winery wastewater for irrigation may also result in K^+ accumulation in the soil, resulting in the leaching of Ca^{2+} and Mg^{2+} and increasing the instability of the soil structure in the long-run (Mosse *et al.*, 2011). Since K^+ has affinity for clay minerals, high soil K^+ can cause clay swelling and dispersion where wastewater is used for irrigation (Arienzo *et al.*, 2012). Similar to Na^+, K^+ in winery wastewater can reduce soil hydraulic conductivity, (Arienzo *et al*., 2009b). However, knowledge regarding negative effects of K^+ on soil structure stability is limited compared to Na^+.

Soil pH tends to increase when wastewater with high pH and Na^+ concentrations is used for irrigation (Lieffering & McLay, 1996). A study carried out in the Western Cape showed that disposal of grape processing effluents changed the soil pH from acidic to alkaline (Papini, 2000). This pH increase was attributed to initial soluble organic matter removal through volatilization of CO_2 during biodegradation.

In contrast, application of wine vinasse containing high bicarbonate slightly reduced the pH of a Mediterranean soil (Bueno *et al.*, 2009). The pH reduction was attributed to the high electrical conductivity of the soil solution (EC_e), *i.e.* 9.2 dS/m, and transformation of organic sugars by micro-organisms. These contrasting results of various studies imply that soil reactions to the application of winery wastewater cannot easily be predicted. In this regard, it is also possible that phosphorus (P) applied via winery wastewater irrigation could contribute towards the nutrient requirements of agricultural crops. The solubility of phosphate (PO_4^{3-}) compounds, or P availability to plants (P is adsorbed by plants in the ionic form $H_2PO_4^-$), strongly depends on the soil pH (Sharpley *et al.*, 1988; Conradie, 1994; Busman *et al*., 2002; Devau *et al.*, 2009). In acidic soils, particularly where pH (Water) is less than 5.5, aluminium (Al^{3+}) and iron (Fe^{3+}) will react with PO_4^{3-} to form amorphous phosphates (Busman *et al*., 2002).

The amorphous Al^{3+} and Fe^{3+} phosphates gradually change to insoluble PO_4^{3-} compounds that are not available to plants. Phosphate also becomes increasingly insoluble if the soil $pH_{(KCl)}$ exceeds 7 (Conradie, 1994; Busman *et al.*, 2002). In alkaline soils, *i.e.* pH > 7, calcium (Ca^{2+}) is the dominant cation that will react with PO_4^{3-}, to form a general sequence of calcium phosphates, *i.e.* dibasic calcium phosphate dihydrate, octocalcium phosphate and hydroxyl apatite (Busman *et al*., 2002). The formation of each of these compounds decreases the solubility of phosphate. On the other hand, PO_4^{3-} solubility can also increase in high pH soils when exchangeable sodium (Na^+) releases inorganic PO_4^{3-} (Sharpley *et al*., 1988). When Na^+ replaces exchangeable Ca^{2+}, Mg^{2+} and Al^{3+} the negative potential of the surface increases, which results in desorption of PO_4^{3-} (Naidu & Rengasamy, 1993). The soils of the South African winelands are highly heterogeneous and can show a high degree of spatial variation in a relatively small area. Soils range in parent material, texture, structure, drainage, coarse fragment content and chemistry. Parent material is usually largely responsible for the physical and chemical makeup of a soil (Van Schoor, 2001b). In the Stellenbosch region, two of the dominant parent materials are shale and granite, while in the Breede River and Olifants River wine growing regions, transported aeolian or fluvial sands are important parent materials (Bargmann, 2003). Due to the heterogeneity of the winelands soils, they are likely to respond differently to the application of winery wastewater, however, there has been little work done to determine these responses.

The objective of this study was to determine the effects of irrigation with diluted winery wastewater on selected chemical properties of four soils varying in parent material and clay content.

5.2. MATERIALS AND METHODS

5.2.1. Soil collection

Details of the pot experiment, wastewater dilution and the irrigation system was described in Chapter 4).

5.2.2. Water sampling and analyses

Water samples were collected prior to each irrigation. The COD in the water was measured using a portable spectrophotometer (Aqualitic COD-reactor®, Dortmund) and the appropriate test kits (COD, CSB, 0-15000 mg/L). The pH and electrical conductivity (EC) were determined according to methods described by Clesceri *et al.* (1998) by a commercial laboratory (Bemlab, Strand). The water was analysed for Ca^{2+}, Mg^{2+}, K^+ and Na^+ by means of atomic emission using an optical emission spectrometer (Varian ICP-OES) at a commercial laboratory (BEMLAB, Strand). Total alkalinity was determined through titration with 0.05N hydrochloric acid. The sodium adsorption ratio (SAR) of the water was calculated as follows (units in meq.L^{-1}):

$$SAR = Na^+ \div [(Ca^{2+} + Mg^{2+}) \div 2]^{½} \qquad \text{(Eq. 5.1)}$$

5.2.3. Soil sampling and analyses

To make provision for destructive soil sampling, each experiment “plot” consisted of four pots. Following each simulated irrigation season, the soil in one of the pots was collected for sampling, *i.e.* after 6, 12, 18 and 24 irrigations. Soil samples were collected from the 0-10 cm and 10-20 cm layers in the pots of all replications. Soil samples were air dried and passed through a 2 mm mesh sieve. All analyses were carried out by a commercial laboratory (Bemlab, Strand). The $pH_{(KCl)}$ was determined in a 1 M potassium chloride (KCl) suspension. The Ca^{2+}, Mg^{2+}, K^+ and 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). Phosphorus was determined according to the Bray2 method, *i.e.* extraction with 0.03 M NH_4F (ammonium-fluoride) in 0.01 M HCl (hydrochloric acid). The P concentration in the extract was determined by inductively coupled plasma optical emission spectrometry (ICP-OES) using a spectrometer (PerkinElmer Optima 7300 DV, Waltham, Massachusetts). For this study, the cations will be referred to as extractable calcium (Ca^{2+}_{extr}), magnesium (Mg^{2+}_{extr}), potassium (K^+_{extr}) and sodium (Na^+_{extr}). The extractable potassium percentage (EPP') was calculated as follows:

$$EPP' = (K^+_{extr} \div S) \times 100 \qquad \text{(Eq. 5.2)}$$

where K^+_{extr} is the extractable potassium (cmol$^{(+)}$/kg) and S is the sum of basic cations (cmol$^{(+)}$/kg). The extractable sodium percentage (ESP') was calculated in the same way to obtain an indication of the sodicity status.

5.2.4. Statistical procedures

Each soil/water treatment was replicated three times in a complete randomised design. The four soils were randomly allocated within each block. The treatment design was a split-plot with soil type as the main plot factor, and soil depth as the sub-plot factor. Analyses of variance were performed separately for each season using SAS version 9.2 (SAS, 2008). The Shapiro-Wilk test was performed to test for non-normality (Shapiro & Wilk, 1965). Student's "t" least significant difference (LSD) was calculated at the 5% significance level to facilitate comparison between treatment means (Ott, 1998). Linear regressions were calculated using STATGRAPHICS® version XV (StatPoint Technologies, Warrenton, Virginia, USA).

5.3. RESULTS AND DISCUSSION

5.3.1. Soil characterization

Soils selected for this study were chosen because they represent dominant soils of the Western Cape wine region. Furthermore, it was expected that the impacts of winery wastewater on soils would differ widely between differently textured soils. The Rawsonville soil was formed from the alluvial gravels of the Breede River. The soils selected for this study, showed no clear stratification and contained a mottled subsoil thereby classifying as a Longlands soil form (orthic A-E horizon - soft plinthic B horizon). The topsoil texture of the soil was fine sand. The soil was slightly acidic with $pH_{(KCl)}$ of 5.7. The geology of the Lutzville region is dominated by metamorphic rocks of the Nama Group in the north and sedimentary rocks of the Cape Super group in the southern and south-western parts (Department of Water Affairs, 2011). However, the soils in this area are mainly derived from aeolian deposited sand (Saayman & Conradie, 1982). The soil was classified as the Garies form (orthic A - Red apedal B horizon – with dorbank as the underlying material). The topsoil texture was fine sand and the soil was neutral with $pH_{(KCl)}$ of 6.7. The Stellenbosch shale soil, was located on the foot hills of Simonsberg mountain. The lower subsoil was derived *in situ* from shale, however, the upper subsoil and A horizon were derived from colluvial material of shale origin. The soil was classified as a red Oakleaf soil form (orthic A – red neocutanic B horizon - unspecified material). The topsoil texture was a fine sandy clay loam and the soil was acidic with $pH_{(KCl)}$ of 4.2. The Stellenbosch granite soil was also located on the foot hills of Simonsberg mountain. The subsoil was derived *in situ* from granite, however the A and E horizons were derived from granitic colluvium. The soil was classified as a Cartref form (orthic A - E horizon - lithocutanic B horizon). Both the A and E horizons were highly leached and hard setting. The topsoil texture was coarse sandy loam. The soil was acidic with $pH_{(KCl)}$ of 4.4.

5.3.2. Chemical composition of the water and amount of elements applied

The mean COD levels in the municipal water and diluted winery wastewater were, 28±4 and 3210±43 mg/L, respectively, during the four simulated seasons (Mulidzi *et al.*, 2016). The COD in the diluted winery wastewater was reasonably close to the target level of 3000 mg/L. As expected, most of the other winery wastewater quality variables were considerably higher compared to the municipal water (Table 4.3). On most irrigation days, the winery wastewater pH was lower compared to the municipal water. The average SAR of the winery wastewater was close to 5 (Table 5.1), which is the limit for irrigation with wastewater according to the South African water quality legislation (Department of Water Affairs, 2013). Due to the differences in the chemical composition of the municipal and diluted winery wastewater, considerably more cations were applied to the soil *via* the wastewater compared to the municipal water (Table 5.2). Total irrigation amounts applied to the Rawsonville sand (1156 mm), Lutzville sand (1126 mm) and Stellenbosch shale (987 mm) over four simulated seasons were comparable, but the Stellenbosch granite (728 mm) received substantially less water. According to Mulidzi *et al.* (2016), this particular soil had a lower water holding capacity and high coarse sand content compared to the other three soils.

Table 5.1. Quality characteristics of municipal water and winery wastewater used for irrigation of four different soils.

Water quality variables	Season				
	1	2	3	4	Mean
			Municipal		
pH KCl	7.7	7.5	7.7	6.9	7.4
EC (mS/m)	8.3	7.2	9.5	9.3	8.6
K^+ (mg/L)	0.8	0.7	0.9	1.6	1.0
Na^+ (mg/L)	7.4	7.2	8.1	8.5	7.8
Ca^{2+} (mg/L)	6.3	6.0	6.1	5.3	5.9
Mg^{2+} (mg/L)	1.3	1.1	1.5	1.8	1.4
SAR	0.7	0.7	0.8	0.8	0.8
HCO_3^-	32.6	22.4	18.4	26.0	24.9
			Winery		
pH KCl	5.3	6.0	4.9	5.6	5.4
EC (mS/m)	94.2	109.8	94.6	119.0	104.4
K^+ (mg/L)	196.1	186.6	204.9	196.4	196.0
Na^+ (mg/L)	75.5	114.9	78.7	68.6	84.4
Ca^{2+} (mg/L)	14.1	18.0	20.0	22.4	18.6
Mg^{2+} (mg/L)	4.9	8.4	6.5	9.1	7.2
SAR	4.5	5.6	4.0	4.1	4.6
HCO_3^-	511.3	655.1	438.2	552.9	539.4

Table 5.2. Amount of elements applied per simulated irrigation season *via* municipal water and diluted winery wastewater, to four different soils.

Element	Season	Amount applied (kg/ha)							
		Rawsonville		Lutzville		Stellenbosch shale		Stellenbosch granite	
		Municipal	Winery	Municipal	Winery	Municipal	Winery	Municipal	Winery
K^+	1	11	3414	11	3312	10	2895	7	2124
	2	12	2535	12	2472	10	2181	7	1587
	3	16	3538	15	3463	13	3034	10	2253
	4	29	3406	28	3312	24	2887	18	2157
Na^+	1	100	1315	97	1276	85	1115	62	818
	2	125	1514	121	1477	107	1303	78	948
	3	139	1358	136	1329	119	1165	89	865
	4	147	1189	143	1156	125	1008	93	753
Ca^{2+}	1	86	245	84	237	73	207	54	152
	2	104	270	101	263	89	232	65	169
	3	106	345	103	338	91	296	67	220
	4	92	388	90	378	78	329	59	246
Mg^{2+}	1	17	85	16	83	14	72	10	53
	2	19	114	18	112	16	98	12	72
	3	26	112	25	110	22	96	17	71
	4	32	158	31	153	27	134	20	100

5.3.3. Potassium and EPP'

Where municipal water was applied, K^{+}_{extr} amounted to 0.21 $cmol^{(+)}$/kg, 0.42 $cmol^{(+)}$/kg, 0.35 $cmol^{(+)}$/kg and 0.31$cmol^{(+)}$/kg, respectively for the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite after the four seasons (data not shown). Since these values were comparable to the baseline values (Table 5.3), it indicated that municipal water irrigation had no effect on the K^{+}_{extr}, irrespective of clay content. In contrast, irrigation with the diluted winery wastewater increased K^{+}_{extr} substantially over the four seasons. The K^{+}_{extr} in the 0-10 cm soil layer was slightly higher compared to the 10-20 cm layer, irrespective of clay content (Fig. 5.1). According to Arienzo *et al*. (2009b), a higher amount of exchangeable K^{+} is retained by soils higher in clay content than soils low in clay content following winery wastewater irrigation. Furthermore, K^{+}_{extr} in the four soils increased linearly with the cumulative amount of K^{+} applied *via* the irrigation water (Fig. 5.1).

In the 0-10 cm layers, the degree of K^{+} extraction was similar for the four soils with an increase of 0.0002 $cmol^{(+)}$/kg per kg K^{+} applied. After the four seasons, EPP' amounted to 4.6%, 11.5%, 13% and 9.5%, respectively, for the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite soils where municipal water was applied (data not shown). Similar to K^{+}_{extr}, EPP' values were comparable to the baseline values (Table 5.3), indicating that the municipal water irrigation did not affect EPP'. In contrast, irrigation with the diluted winery wastewater increased EPP' over the four seasons (Fig. 5.2). The EPP' in the 0-10 cm soil layer was slightly higher compared to the 10-20 cm layer, with the exception of Stellenbosch granite soil. In the case of the sandy soils and Stellenbosch shale soil, the EPP' in the 0-10 cm showed a slower increase following the second season (Figs. 5.2A, 5.2B & 5.2C).

The EPP' in the 10-20 cm layer showed an almost linear increase with applied K^{+}. In the case of Stellenbosch granite, these trends did not occur as EPP' was comparable in both soil layers (Fig. 5.2D). After the fourth season, EPP' was similar in both layers which suggested that the granite soil was no longer retaining high amounts of K^{+} in the 0-10 cm layer.

Table 5.3. Initial extractable cations, extractable potassium percentage (EPP'), extractable sodium percentage (ESP') and $pH_{(KCl)}$ in the four soils selected for the study.

Variable	Rawsonville sand	Lutzville sand	Stellenbosch shale	Stellenbosch granite
K^{+}_{extr} ($cmol^{(+)}$/kg)	0.2	0.5	0.4	0.3
Na^{+}_{extr} ($cmol^{(+)}$/kg)	0.1	0.1	0.1	0.2
EPP'	3.7	13.2	13.8	9.7
ESP'	1.9	2.6	3.4	6.5
Ca^{2+}_{extr} ($cmol^{(+)}$/kg)	3.5	2.4	1.6	1.8
Mg^{2+}_{extr} ($cmol^{(+)}$/kg)	1.6	0.8	0.8	0.8
$pH_{(KCl)}$	5.7	6.7	4.2	4.4

For healthy grapevine growth in soils with a pH below 6, it is recommended that a K^+ saturation of 4% is required on the exchange sites (Conradie, 1994). Prior to irrigation, the EPP' was greater than 4% in all soils, except for the Rawsonville sand which had an EPP' of 3.7%, which was close to the threshold (Table 5.3). Thus for the soils investigated, K^+ added *via* the wastewater does not represent a benefit in terms of nutrient balance and supply. In fact, high K^{+}_{extr} levels may cause excessive absorption by grapevines which could result in high wine pH, and eventually reduce colour stability of red wines where winery wastewater is applied (Mpelasoka, 2003; Kodur, 2011).

Under normal cropping conditions, there is a possibility that K^+ applied *via* wastewater can be beneficial if it can maintain optimum levels when K^+ is absorbed by grapevines and/or inter-row crops, or if K^+ is leached by rainfall in winter. It should be noted that the observed K^+ accumulation occurred in the absence of rainfall or crops. Determining the effect of leaching by winter rainfall where diluted winery wastewater is used for irrigation, is part of a separate study.

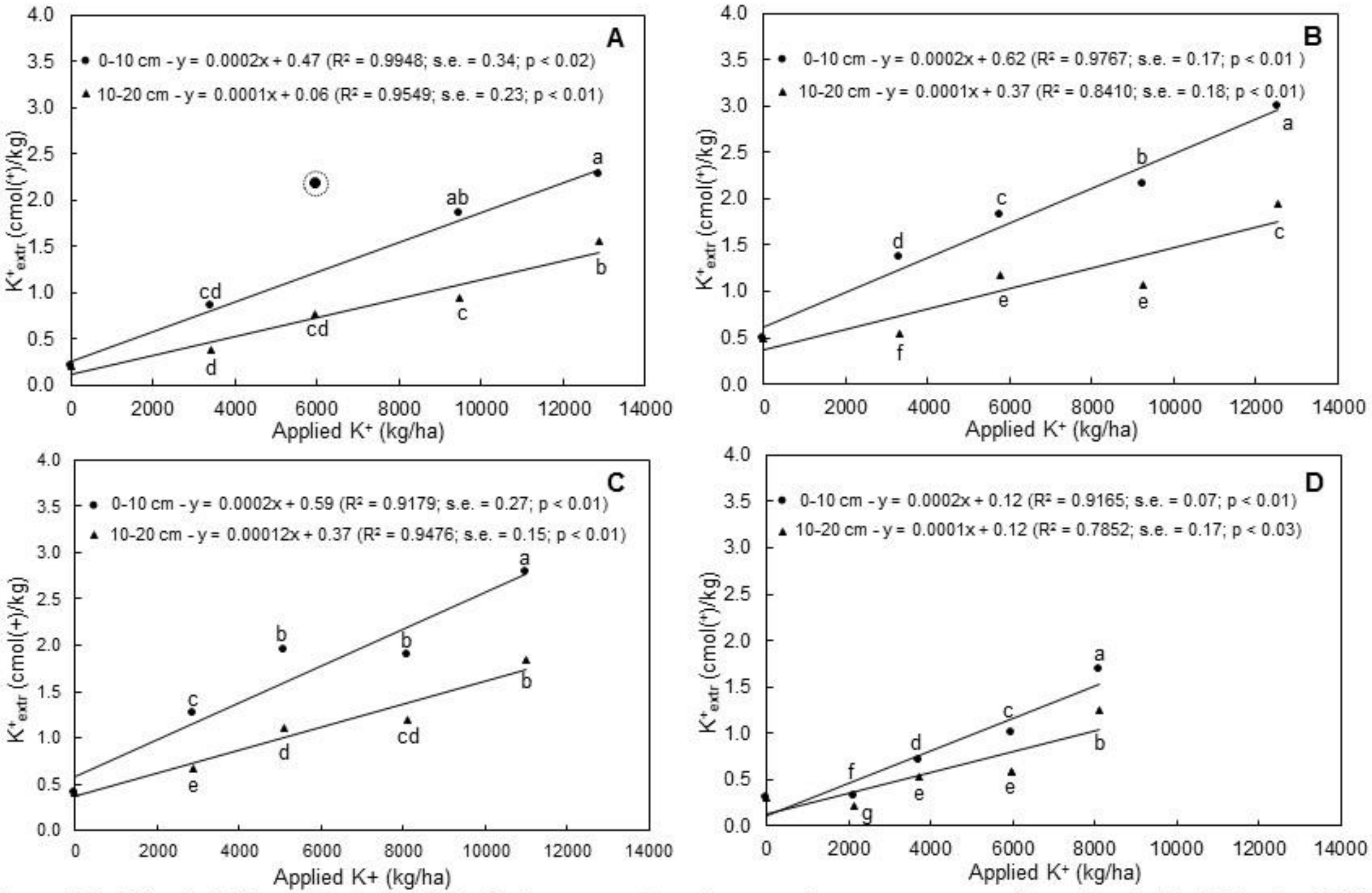

Figure 5.1. Effect of K^+ applied *via* diluted winery wastewater over four seasons on the extractable K^+ in the 0-10 cm and 10-20 cm layers of (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils. The encircled data point was regarded as an outlier due experimental error and were not included in the equation. Values designated by the same letter do not differ significantly ($p \leq 0.05$).

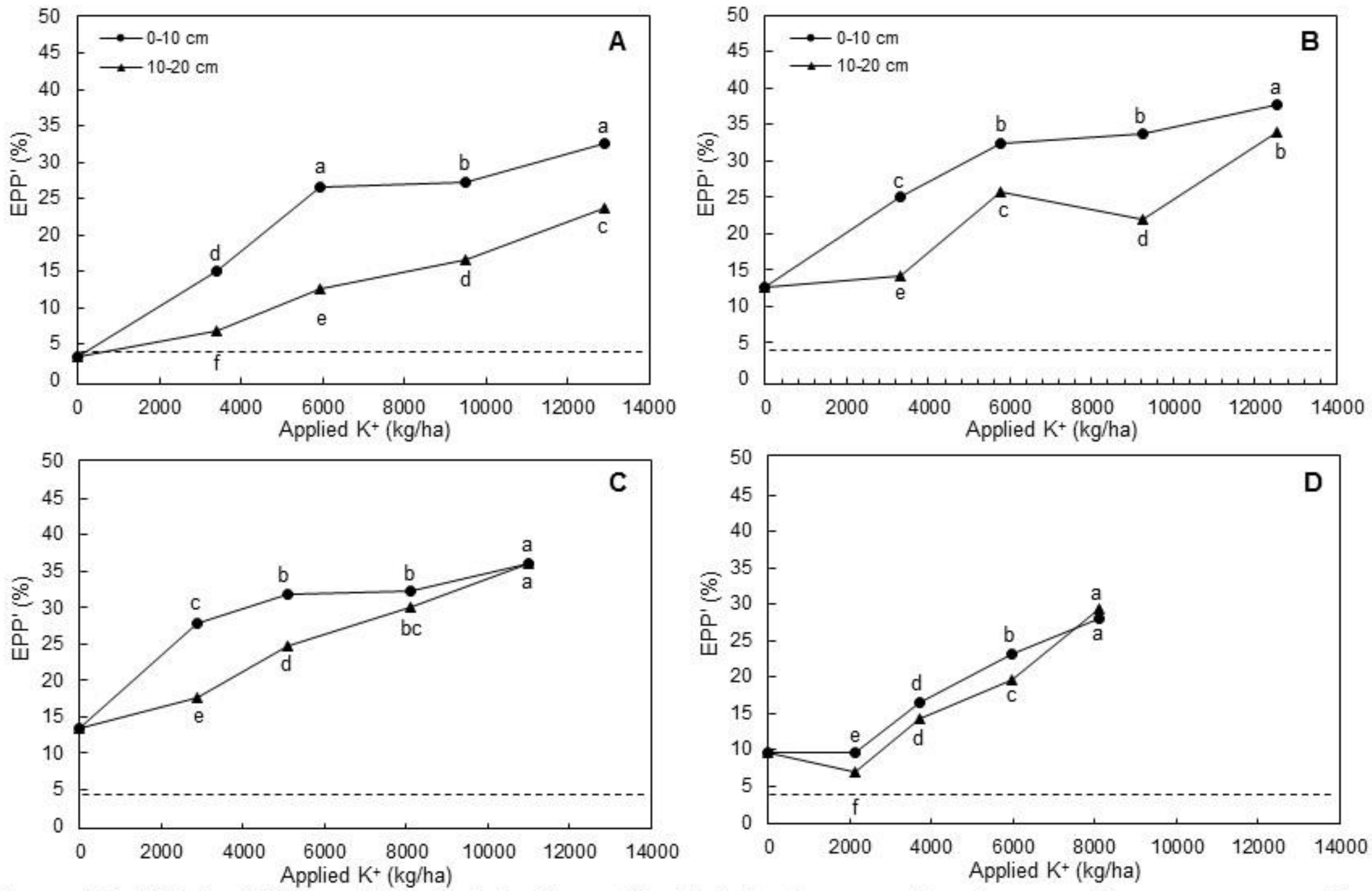

Figure 5.2. Effect of K⁺ applied *via* irrigation with diluted winery wastewater over four seasons on the extractable potassium percentage (EPP') in the 0-10 cm and 10-20 cm layers of (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils. The dashed line indicate the critical EPP' threshold for grapevines. Values designated by the same letter do not differ significantly ($p \leq 0.05$).

5.3.4. Sodium and ESP'

Where municipal water was applied, Na^{+}_{extr} amounted to 0.15 $cmol^{(+)}$/kg, 0.17 $cmol^{(+)}$/kg, 0.16 $cmol^{(+)}$/kg and 0.25 $cmol^{(+)}$/kg, respectively, for the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite soils after the four seasons (data not shown). Being comparable to the baseline values (Table 5.3), it indicated that municipal water irrigation had almost no effect on the Na^{+}_{extr}, irrespective of clay content. On the other hand, irrigation with the diluted winery wastewater increased Na^{+}_{extr} substantially over the four seasons. In all the soils, the degree of Na^{+}_{extr} accumulation in the 0-10 cm layer was higher compared to the 10-20 cm layer (Fig. 5.3). The difference between the layers was most prominent in the shale followed, by the granite and sandy soils (Figs. 5.3C & 5.3D). These trends indicated that more Na^{+} was extracted in the 0-10 cm layer of the heavier soils compared to the sandy soils. The increased extraction of Na^{+} from the top layer, may be as a result of less sorption of Na^{+} to the soil and evaporative concentration of Na^{+} in the evaporating soil solution. In fact, previous studies have shown that the adsorption of Na^{+} was reduced by the presence of high K^{+} levels where winery wastewater was applied (Laurenson *et al.*, 2012 and references therein).

In all soils, the Na^{+}_{extr} increased linearly with the cumulative amount of Na^{+} applied *via* the irrigation water (Fig. 5.3). However, the rate of increase in Na^{+}_{extr} with increase in applied Na^{+} ($Na^{+}_{extr}/Na^{+}_{appl}$) differed between the soils. The $Na^{+}_{extr}/Na^{+}_{appl}$ increased with clay content in the 0-10 cm layer, but no correlation was observed in the 10-20 cm layer (Fig. 5.4). Where municipal water was applied, the ESP' amounted to 3.2%, 4.4%, 2.9% and 4.3%, respectively, in the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite soils after four seasons. The ESP' values were comparable to the baseline values with the exception of Stellenbosch granite soil that had a higher baseline ESP' (Table 5.3). Where winery wastewater was applied over four seasons, the ESP' did not show a definite linear increase with the amount of Na^{+} applied in any of the layers (Fig. 5.5).

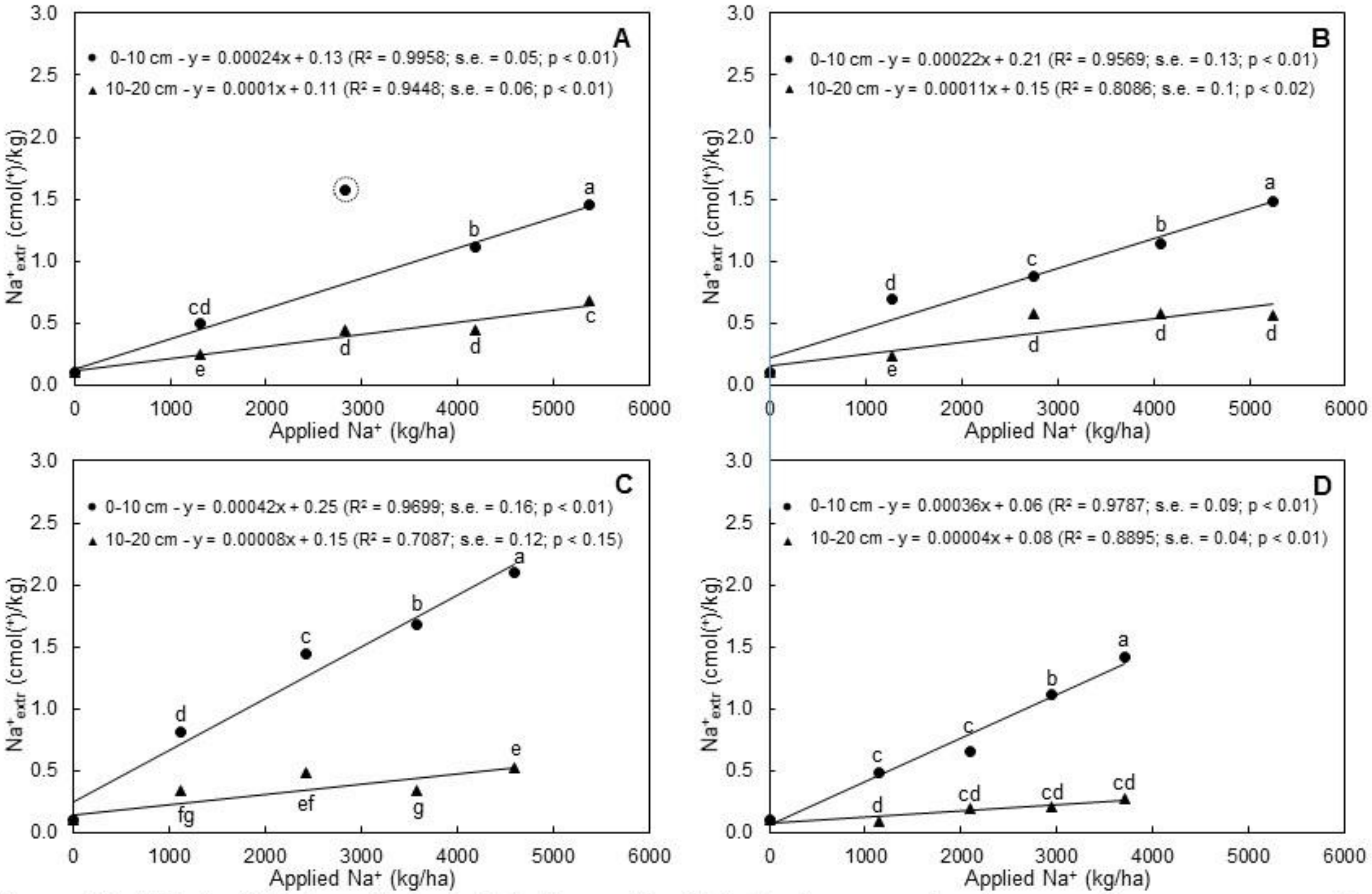

Figure 5.3. Effect of Na^+ applied *via* irrigation with diluted winery wastewater over four seasons on the extractable Na^+ in the 0-10 cm and 10-20 cm layers of (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils. The encircled data point was regarded as an outlier due experimental error and was not included in the equation. Values designated by the same letter do not differ significantly ($p \leq 0.05$).

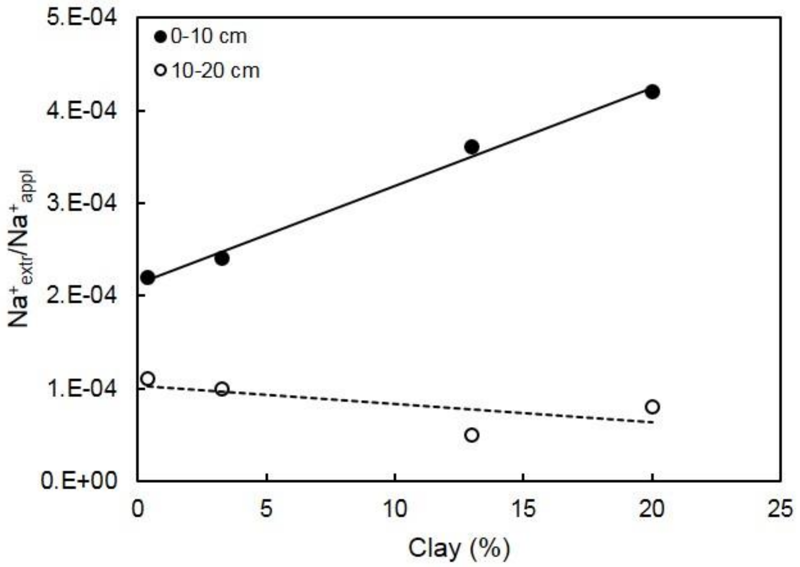

Figure 5.4. Relationship between the ratio of extractable sodium (Na^+_{extr}) to sodium applied per hectare (Na^+_{appl}) and clay content for four different soils.

In the case of the Rawsonville sand, the ESP' exceeded the critical threshold of 15% for sustainable agricultural use from the second season onwards in the 0-10 cm layer (Fig. 5.5A). Wastewater irrigation increased the ESP' above 15% after the second season in the Lutzville sand, but also only in the 0-10 cm layer (Fig. 5.5B). From the first season, the ESP' exceeded 15% only in the 0-10 cm layer of the Stellenbosch shale soil (Fig. 5.5C). Although no infiltration problems occurred after four seasons, it does not rule out the possibility that sodicity could have negative effects on soil structure in the long run. In the case of the Stellenbosch granite soil, the ESP' exceeded 15% after the third season, but also only in the 0-10 cm layer (Fig. 5.5D). Although the ESP' in the two sandy soils seemed to have reached a plateau at c. 20%, it might induce negative effects on grapevine growth and yield if the ESP' remains near the threshold over time. Given the higher ESP' in the heavier soils, sodicity will have negative effects on plant growth and soil physical conditions if these soils are irrigated with winery wastewater, even when diluted. The Stellenbosch shale soil showed no visual signs of infiltration problems but water infiltration into the Stellenbosch granite soil was considerably slower where the wastewater was applied compared to the municipal water.

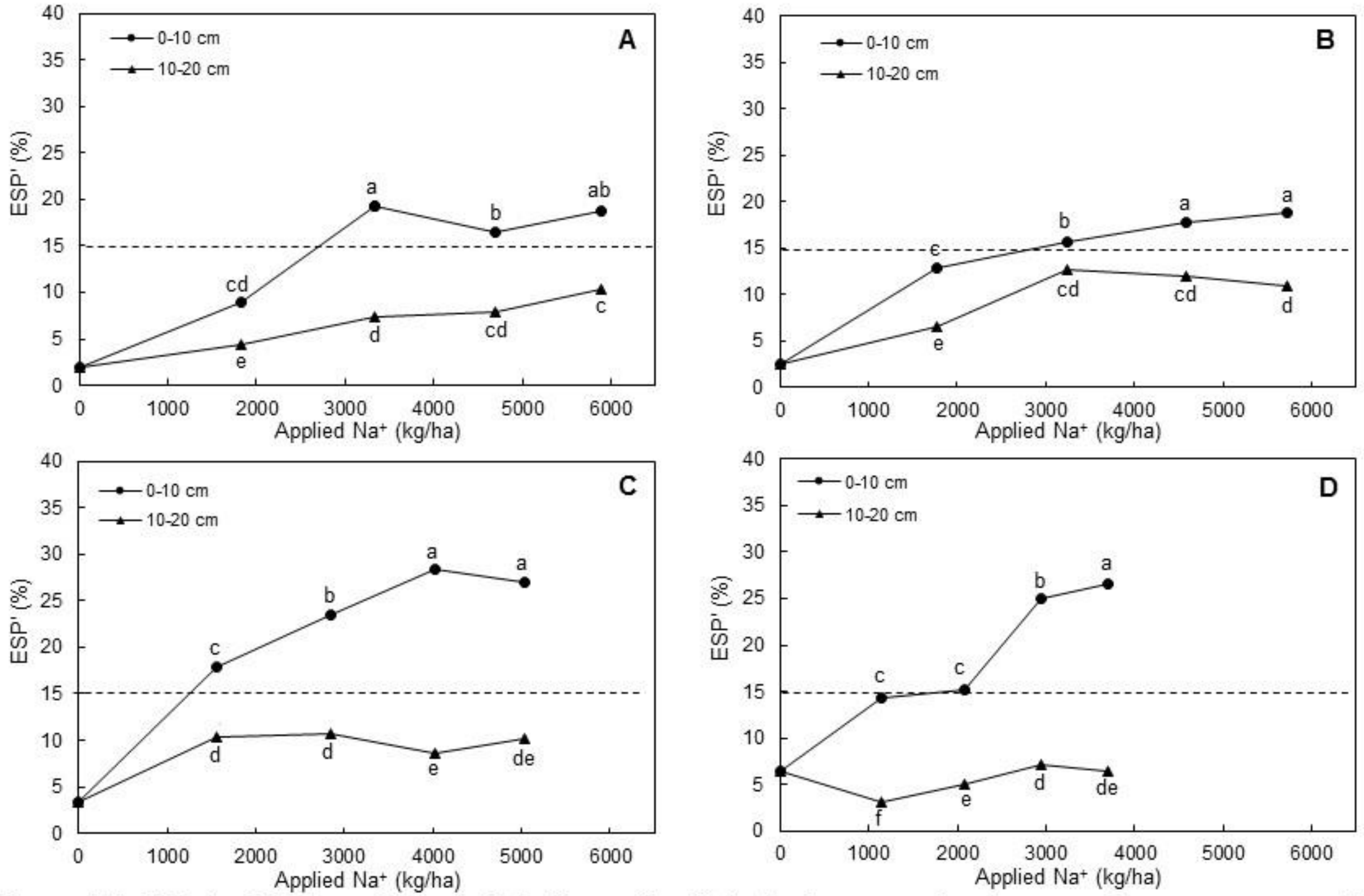

Figure 5.5. **Effect of Na⁺ applied *via* irrigation with diluted winery wastewater over four seasons on the extractable sodium percentage (ESP') in the 0-10 cm and 10-20 cm layers of (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils. The dashed line indicate the critical ESP' threshold for grapevines. Values designated by the same letter do not differ significantly ($p \leq 0.05$).**

It must be noted that the infiltration problems occurred right from the first season, *i.e.* when the ESP' in the top layer was around 15% (Mulidzi *et al*., 2016). It is well documented that Ca^{2+} and Mg^{2+} can counter the negative effects of Na^{+} on water infiltration but Ca^{2+}_{extr} and Mg^{2+}_{extr} in the Stellenbosch shale and granite soils were comparable (Table 5.3). It was previously reported that the saturated conductivity of a topsoil of a similar granitic soil at Nietvoorbij was 112 mm/h (Myburgh, 2015). Since the drip application rate was 115 mm/h (Mulidzi *et al.*, 2016), it could be that the infiltration rate of the granitic soil was exceeded, thereby causing the slow water infiltration. Another possible reason for the slow infiltration rate in the granitic soil is the dispersive nature of the bleached topsoil. Bleached topsoils are pale in colour due to the loss of Fe^{2+} from the horizon. Iron oxides play an important role in stabilizing clays against dispersion (Tombacz *et al*., 2004). The lack of Fe^{2+} in the granitic topsoil, might make this soil more susceptible to clay dispersion and surface sealing when irrigated with wastewater containing high levels of Na^{+} and K^{+}. The red Oakleaf soils in the Stellenbosch region have a high Fe^{2+} content (Le Roux, 2015). This may explain why infiltration in the Stellenbosch shale was unhindered despite the poor quality of the irrigation water.

5.3.5. Calcium and magnesium

After the four simulated irrigation seasons, Ca^{2+}_{extr} and Mg^{2+}_{extr} did not show any trends that could be related to the amounts of these elements applied *via* the municipal water and diluted winery wastewater, respectively (Table 5.4). The lack of response was probably due to the small amounts of Ca^{2+} and Mg^{2+} applied through the irrigation water (Table 5.2). In fact, irrigation with the wastewater reduced the Ca^{2+}_{extr} in the Rawsonville sand after the four seasons. The Mg^{2+}_{extr} in the Lutzville sand showed a similar trend (Table 5.4). Where wastewater was applied to the Stellenbosch granite soil, Mg^{2+}_{extr} was also lower compared to Mg^{2+}_{extr} in the 0-10 cm layer of the municipal water irrigation. The foregoing implied that irrigation with winery wastewater is unlikely to have any benefits in terms of Ca^{2+} and Mg^{2+} supply to plants. Furthermore, if applied in such small amounts, these elements will not be able to counter possible structural problems caused by high levels of Na^{+} applied *via* winery wastewater.

Table 5.4. Effect of irrigation with municipal water and diluted winery wastewater on the extractable Ca^{2+} and Mg^{2+} in four different soils after four simulated seasons.

Soil	Municipal		Winery	
	0-10 cm	10-20 cm	0-10 cm	10-20 cm
	Ca^{2+}_{extr} (cmol$^{(+)}$/kg)			
Rawsonville sand	3.5a[1]	3.5a	3.1b	3.1b
Lutzville sand	2.7a	3.1a	2.9a	2.7a
Stellenbosch shale	1.9a	1.7a	2.0a	1.9a
Stellenbosch granite	2.4a	2.2a	2.1a	1.9a
	Mg^{2+}_{extr} (cmol$^{(+)}$/kg)			
Rawsonville sand	1.3a	1.4a	1.2a	1.2a
Lutzville sand	0.8a	0.7b	0.6c	0.5d
Stellenbosch shale	0.8a	0.7b	0.9a	0.9a
Stellenbosch granite	1.0a	0.5d	0.9b	0.7c

[1] Values designated by the same letter within each row do not differ significantly ($p \leq 0.05$).

5.3.6. pH$_{(KCl)}$

The pH$_{(KCl)}$ of the soils prior to any treatment is given in Table 5.3. The Stellenbosch soils had a low pH$_{(KCl)}$ (4.2-4.4) while the Lutzville and Rawsonville sands were substantially higher (6.7 and 5.7, respectively). Where municipal water was applied, soil pH$_{(KCl)}$ was 5.9, 7.4, 4.5 and 4.6, respectively, for the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite soils after the four seasons (data not shown). This indicated irrigation with municipal water did not substantially affect pH$_{(KCl)}$, irrespective of soil clay content (Table 5.3). In contrast, irrigation with diluted winery wastewater increased pH$_{(KCl)}$ substantially in all the soils over the four seasons (Fig. 5.6). In all the soils, pH$_{(KCl)}$ in the 0-10 cm soil layers tended to be higher compared to the 10-20 cm layer. This means that despite the wastewater having a fairly low pH (4.9-6.0) it actually increased the soil pH. The Lutzville, Rawsonville and Stellenbosch shale soils showed a pH increase of approximately 2 pH units, while the granite soil, which received less irrigation water only showed a pH increase of 1 unit. Although this may seem counter intuitive it is not an unusual phenomenon and has been recorded in numerous studies where organic substrates are added to a soil (Yan *et al*., 1996; Li *et al*., 2008; Rukshana *et al*., 2011; Rukshana *et al*., 2012).

When salts of organic acids are added to a soil, decarboxylation and hydrolysis of the organic/bicarbonate anions increases the pH (Li *et al.,* 2008). The winery wastewater used in this study has an extremely high total alkalinity (Table 5.1). It is likely that this alkalinity comprises of a number of deprotonated organic acids as well as bicarbonate ions. The charge on these anions is largely countered by K^+ and Na^+ cations, thus when applied to soils this results in a pH increase due to decarboxylation and anion hydrolysis reactions as described by Li *et al*. (2008). These authors found that Na^+ and K^+ organic salts are more effective at increasing soil pH than Ca^{2+} and Mg^{2+} organic salts. This would explain why the soil $pH_{(KCl)}$ increased linearly with the cumulative amount of K^+ plus Na^+ applied *via* the diluted winery wastewater (Fig. 5.6). Similar increases in pH were reported by Laurenson *et a*l. (2012) when high alkalinity winery wastewater was applied to vineyard soils.

Initially, $pH_{(KCl)}$ in the Rawsonville and Lutzville sands (Table 5.3) was higher than the lower threshold of 5.5 for vineyard soils (Conradie, 1994). However, where these soils were irrigated with diluted winery wastewater, the high $pH_{(KCl)}$ levels (Fig. 5.6) could have detrimental effects on the availability of plant nutrients (Busman *et al.,* 2002). Where the $pH_{(KCl)}$ was initially lower than 5.5 in the Stellenbosch shale and granite soils, irrigation with the diluted winery wastewater had a beneficial effect by raising the $pH_{(KCl)}$ to the optimum range after the first season (Fig. 5.6C and D). In sandy soils where the pH is not well buffered, vineyard soils may become acidic under intensive irrigation, particularly drip irrigation (Myburgh, 2012b). Such soils, *e.g.* the sandy vineyard soils in the Olifants River region, require frequent liming. Therefore, irrigation with diluted winery wastewater containing high levels of K^+ may reduce the rate of acidification in these poorly buffered sandy soils.

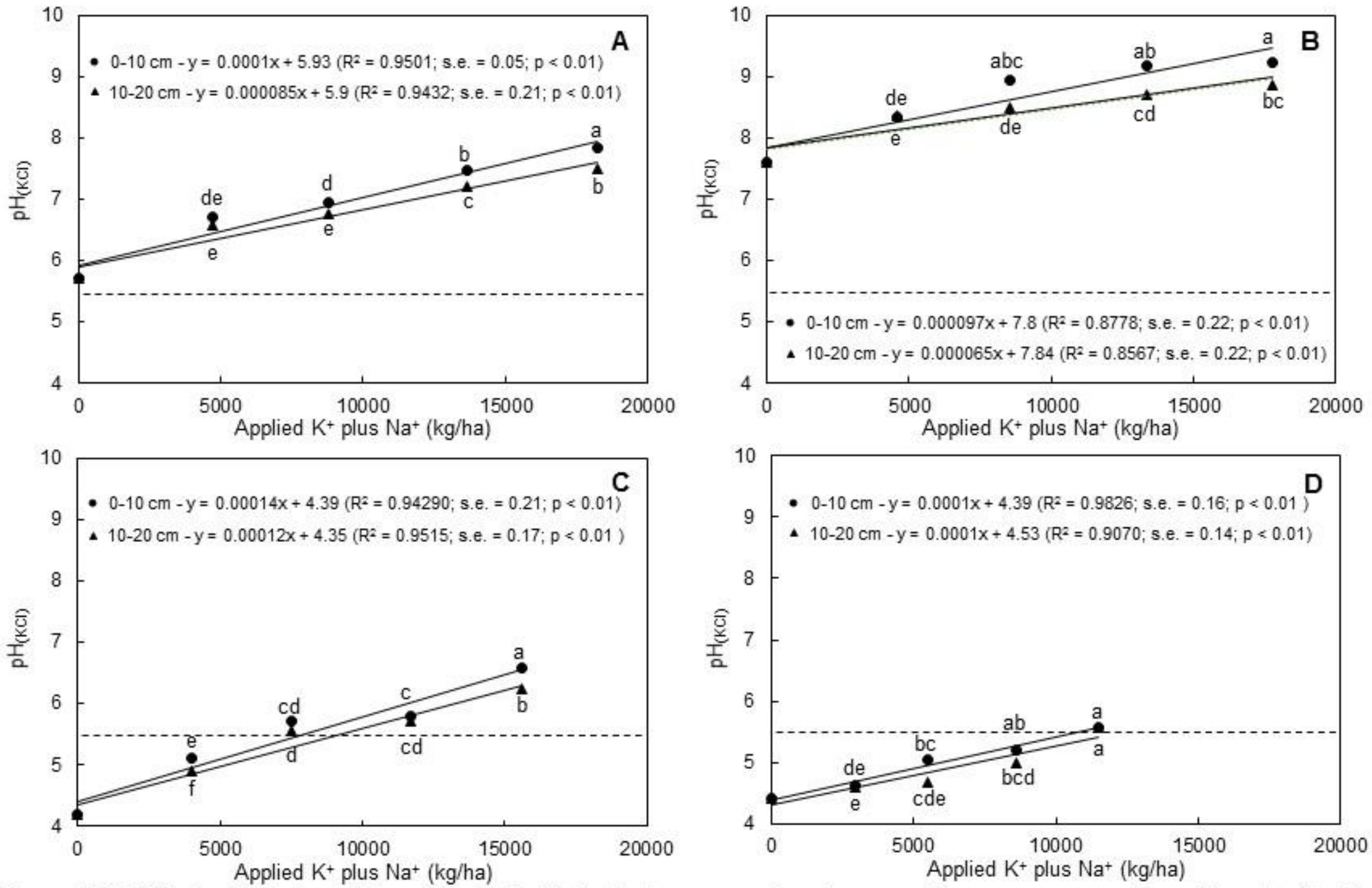

Figure 5.6. Effect of K^+ plus Na^+ applied *via* diluted winery wastewater over four seasons on the $pH_{(KCl)}$ in the 0-10 cm and 10-20 cm layers of (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils. Dashed lines indicate lower $pH_{(KCl)}$ threshold for grapevines. Values designated by the same letter do not differ significantly ($p \leq 0.05$).

5.3.7. Phosphorus

The initial P contents were 217 mg/kg, 6 mg/kg, 8 mg/kg and 15 mg/kg, in the Rawsonville sand, Lutzville sand, Stellenbosch shale and Stellenbosch granite soils respectively. With the exception of the Rawsonville sand, P contents in the soils were in line with values normally expected for vineyard soils (Conradie, 1994). The initial P levels in the Rawsonville sand were more than ten-fold the maximum of 20 mg/kg recommended for grapevines in soils containing less than 6% clay (Conradie, 1994). It was also more than double the P level at which wheat yields were reduced in a red, sandy soil near Vaalharts (Eloff & Laker, 1978). It was previously reported that P levels could range between 10 and 400 mg/kg for the duplex and gradational soils in Australia (Naidu & Rengasamy, 1993 and references therein). The foregoing confirmed that high levels of P are not uncommon in agricultural soils. Irrigation with municipal water had minimal effect on the P contents in all of the soils (data not shown). The change in extractable P of the four soils after wastewater irrigation is shown in Figure 5.7. The P content in the 10-20 cm layer of the Rawsonville sand only tended to be higher compared to the top layer following the third diluted winery wastewater irrigation, thereby indicating that attenuation of P did not occur in the top layer (Fig. 5.7A).

The drastic decline of available P in the Rawsonville sand during the third season of winery wastewater irrigation (Fig. 5.7A) was possibly due the formation of stable complexes with constituents in the wastewater from which P could not be extracted by the Bray II reagent (Eloff & Laker, 1978). Since no leaching occurred when irrigations were applied (Mulidzi *et al.*, 2016), it could not have contributed to the decline in available P. In contrast, irrigation with diluted winery wastewater increased soil P substantially more in the 0-10 cm layer compared to the 10-20 cm layer of the Lutzville sand and the Stellenbosch granite soil over the four simulated seasons (Fig. 5.7B & 5.7D). This trend indicated that P attenuation occurred in the top layer of these soils. The very large increase in plant-available P in the top layer of the very sandy red soil from Lutzville is striking. It confirms the ability of non-acid red sandy soils 1 to retain applied P in plant-available forms, as reported by others (e.g. Eloff & Laker, 1978). On the one hand there is little movement of P in the soil, but there is on the other hand also little fixation of P into unavailable forms. Available P in the Lutzville sand increased as the pH (KCl) increased well above 7 where the diluted wastewater was applied (Fig.5.7B).

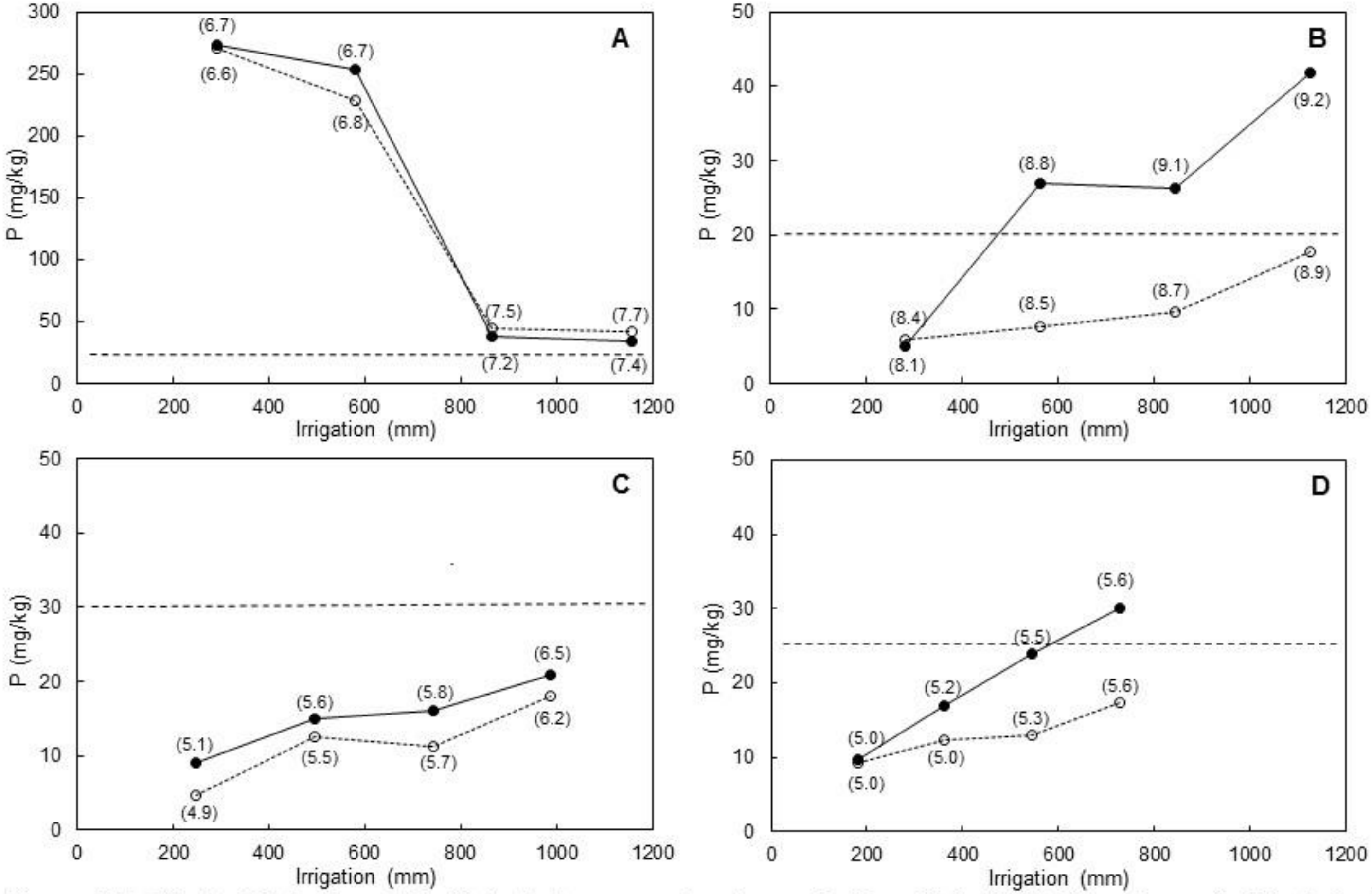

Figure 5.7. Effect of irrigation with diluted winery wastewater on P (Bray 2) in the 0-10 cm layer (solid circles) and 10-20 cm layer (open circles) in (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils over four simulated seasons. Values in brackets indicate the soil $pH_{(KCl)}$. Dashed lines indicate the P (Bray 2) thresholds for grapevines based on clay content (Conradie, 1994).

This trend suggested that the increasing amounts of sodium applied *via* the wastewater increased the soluble $PO4^{3-}$, instead of insoluble calcium phosphates being formed. Although the P content in the 10-20 cm layer of the Stellenbosch shale tended to be lower after the first simulated season, it increased at the same rate over time as in the 0-10 cm layer (Fig. 5.7C). This indicated that no P attenuation occurred from the second season onwards.

In the case of the initially acidic Stellenbosch shale and granite soils (Fig. 5.7C & 5.7D), the amorphous Fe^{3+} and Al^{3+} phosphates became more soluble as the $pH_{(KCl)}$ increased towards the optimum as proposed by Busman *et al.* (2002). Since P was not determined in the irrigation water, models to estimate the effect of irrigation with diluted winery wastewater on soil P based on the amounts applied, could not be created. However, the general variation in available P for the four soils could be illustrated with a plot of relative P, as calculated for each soil and layer, against $pH_{(KCl)}$ (Fig. 5.8).

After the fourth season, available P in the Rawsonville sand was still above the norm of 20 mg/kg proposed by Conradie (1994) for grapevines in sandy soils (Fig. 5.7A). However, this must be regarded as an atypical situation due to the initially high levels. After four simulated seasons of irrigation with the winery wastewater, Bray II P in the Lutzville sand reached over 40 mg/kg, thus far exceeding the norm of 20 mg/kg (Fig. 5.7B). This indicates that the winery wastewater is a good source of P on such soils. On the other hand, the fact that the Bray II P content of this soil increased by nearly 40 mg/kg after four seasons, could serve as a warning that long term continuous application of winery wastewater could cause accumulation of excessive P levels in such soil over time.

After the fourth season, P in the Stellenbosch shale soil (Fig. 5.7C) was well below the norm of 30 mg/kg for grapevines in soils containing more than 15% clay (Conradie, 1994). Likewise, P in the Stellenbosch granite soil (Fig. 5.7D) was less than the lower threshold of 25 mg/kg for soils containing 6% to 15% clay (Conradie, 1994). The much smaller increases in available P in the two Stellenbosch soils indicate much larger P fixation into unavailable forms in these acidic soils than in the Lutzville soil.

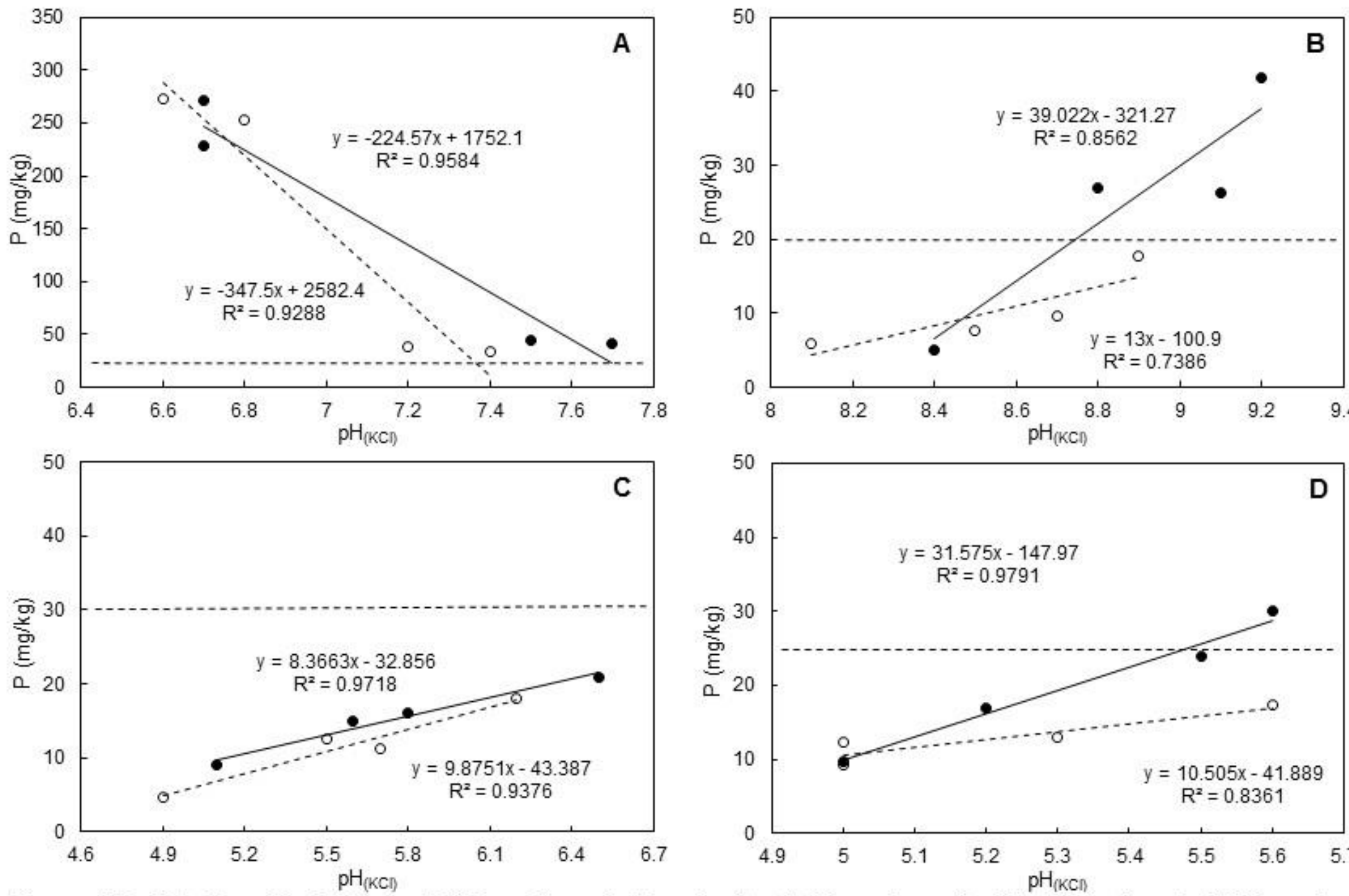

Figure 5.8. Relationship between P (Bray 2) and $pH_{(KCl)}$ in the 0-10 cm layer (solid circles) and 10-20 cm layer (open circles) in (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale and (D) Stellenbosch granite soils following irrigation with diluted winery wastewater over four simulated seasons. Dashed lines indicate the P (Bray 2) thresholds for grapevines based on clay content (Conradie, 1994).

Although the lower thresholds were not reached, it does not rule out the possibility that it could be achieved if diluted winery wastewater is applied over a longer period. However, if the P applied *via* winery wastewater is absorbed by grapevines and cover crops, the minimum thresholds might not be exceeded to the extent that no fertilizers will be required.

5.4. CONCLUSIONS

Irrigation with winery wastewater containing relatively high levels of K^+ and Na^+ affected the soil compared to the municipal water control. Since the K^+_{extr} increase with increasing amounts of K^+ applied was comparable for the four soils, it suggested that clay content did not play a significant role. The EPP' was above the critical level of 4% in all the soils before the experiment commenced.

This means that, under the prevailing conditions, there is a high risk of K^+ accumulating to levels that could have negative effects on wine colour if the excess K^+ is not leached out in winter or absorbed by inter-row crops in summer. In the heavier soils the increase of Na^+_{extr} with increasing amounts of Na^+ applied was almost double compared to the sandy soils. This indicated that the risk of Na^+ reaching excessive levels will be less where vineyards in sandy soils are irrigated with diluted winery wastewater than in heavier soils. Although the ESP' exceeded the threshold of 15% only in the 0-10 cm layer, Na^+ accumulation in the deeper layers could increase ESP' to excessive levels in the long run.

Due to low Ca^{2+} and Mg^{2+} concentrations in the diluted winery wastewater, their extractable concentrations in the soil were comparable to the initial levels after four seasons. This indicated that these elements are not contained in the cleaning detergents used in wineries to the extent that they would accumulate in the soil, irrespective of clay content. The soil $pH_{(KCl)}$ increase irrespective of clay content, could probably be attributed to organic anions added to the soil *via* irrigation with diluted winery wastewater.

Where diluted winery wastewater was applied, the level of soluble P in the shale and granite soils increased. Although the initial $pH_{(KCl)}$ in the aeolic sand was higher than the optimum range, the presence of relatively high levels of Na^+ caused available P to increase as the $pH_{(KCl)}$ increased. In the case of the alluvial sand containing unusually high initial levels of P, the $pH_{(KCl)}$ increased out of the optimum range, thereby causing a substantial reduction in the level of available P. These results indicated that irrigation with diluted winery wastewater could promote P absorption by grapevines if the $pH_{(KCl)}$ shift is towards the optimum. Since the level of P applied *via* diluted winery wastewater appears to be generally low, application of P fertilizers will probably still be necessary to ensure adequate uptake by grapevines. In the sandy soils, where the $pH_{(KCl)}$ approached 8, or even higher values, nutrient solubility and absorption could be reduced if winery wastewater is used for vineyard irrigation. It must be noted that the foregoing results represent a worst case scenario, *i.e.* in the absence of rainfall or crops. Determining the effect of seasonal leaching by winter rainfall on the chemical status in soils irrigated with diluted winery wastewater will be discussed in Chapter 6.

CHAPTER 6. EFFECT OF SIMULATED WINTER RAINFALL ON SELECTED SOILS IRRIGATED WITH WINERY WASTEWATER

6.1. INTRODUCTION

Studies regarding climate change in the wine growing regions of the Western Cape Province have shown sharp increases in air temperature, whereas rainfall is expected to decline, or to be differently distributed during the rainy season (Vink *et al*., 2012). Winter rainfall after winery wastewater irrigation will lead to the leaching of nutrients to the groundwater. Changes in soil structure due to wastewater irrigation depend on the quality of wastewater *i.e.* salinity levels, organic matter content, and the amount of total suspended solids (Muller *et al*., 2007). The rate and amount of pollutants from winery wastewater reaching groundwater resources depend on several factors such as: sorption, degradation, chemical properties of the wastewater, soil characteristics, environmental conditions, rainfall and water management practices (Muller *et al*., 2007). Saline-sodic irrigation water in low rainfall and high evaporation areas will increase soil sodicity (Jalali *et al*., 2008). Furthermore, a major side effect associated with wastewater irrigation is the potential irreversible deterioration of the groundwater quality (US EPA, 2004). The electric conductivity (EC) of Fluvisol soils in Tunisia decreased as a result of leaching of salts by the Autumn-Spring rainfall (Kallel *et al*., 2012). However, due to high cation exchange capacity (CEC) and high water retention capacity, the soils still retain high levels of Na^+. Winter rainfall on soils irrigated with winery wastewater will lead to the reduction of soil electrolyte concentrations regardless of soil type. Low rainfall areas are likely to experience less soil structural hazard which is linked to high exchangeable monovalent cation concentrations while high rainfall areas will experience more soil structural hazard (Suarez *et al*., 2008).

The objective of this study was to investigate the effect of simulated winter rainfall on leaching of basic cations and subsequent pH changes in soils irrigated with diluted winery wastewater.

6.2. MATERIALS AND METHODS

6.2.1. Soils used

Six pedogenetically different soils commonly found in the Western Cape Province were included in the study (Table 6.1). The taxonomic classification of the soils is given according to the South African soil classification system (Soil Classification Working Group, 1991). For the purpose of this study, soils will be referred to as Rawsonville sand, Lutzville sand, Stellenbosch shale, Stellenbosch granite, Stellenbosch sand and Robertson clay.

6.2.2. Soil collection

The sandy Longlands soil was collected in a vineyard near Rawsonville, whereas the sandy Garies soil was collected from open land near Lutzville. The shale derived Oakleaf and granite derived Cartref soils were collected from the Nietvoorbij experiment farm of the Agricultural Research Council (ARC) near Stellenbosch. Detailed descriptions of these four soils, and how they were collected are presented in Chapter 3. The sandy Kroonstad soil was collected from a grass grazing paddock at a winery near Stellenbosch. The area was previously cultivated. Soil was collected at 30 different positions, *i.e.* approximately 10 m apart, in the paddock. The soil was sampled at a depth of 300 mm. The composited samples were put through a 6 mm sieve in order to remove large fragments such as stones. The clayey Valsriver soil was collected from an area which was previously used for lucerne production at the ARC experiment farm near Robertson. It was collected and prepared according to the same procedure as the Kroonstad soil.

6.2.3. Packing of soils in pots to a predetermined bulk density

The procedure for packing the soils into the PVC pots to a specific predetermined bulk density is described in Chapter 4.

6.2.4. Application of water to the soils

For the control treatment, the soils were irrigated with water abstracted from the Holsloot River near Rawsonville in the Breede River valley. Water for the wastewater treatment was collected from the wastewater pit at a winery near Rawsonville. The winery wastewater was then diluted to a chemical oxygen demand (COD) level of 3000 mg/L.

Table 6.1. Origin, Taxonomic and World Reference Base (WRB) classifications, as well as general description and co-ordinates for the six soils used.

Origin	Classification		General description	Co-ordinates
	Taxonomic	WRB		
Rawsonville	Longlands	Gleyic, albic,Arenosol	Alluvial sand, from a vineyard	-33.4137.7° 19.1920.3°
Lutzville	Garies	Eutric, petric, Durisol	Aeolian sand, from open land	-31.5589.1° 18.3531.2°
Stellenbosch	Oakleaf	Chromic, Acrisol	Shale derived, from open land	-33.550.28° 18.520.69°
Stellenbosch	Cartref	Albic, leptic, Acrisol	Granite derived, from open land	-33.5439.9° 18.5216.6°
Stellenbosch	Kroonstad	Gleyic, albic,Planosol	Sandy soil from open land	-33.4958.6° 18.4759.9°
Robertson	Valsrivier	Chromic, Lixisol	Clay soil from cultivated area	-33.4923.6° 19.5236.0°

Irrigations were applied over one simulated season, which consisted of six irrigations. It was estimated that six is the number of irrigations a micro-sprinkler irrigated vineyard would require during the harvest period, *i.e.* when the highest volumes of wastewater are produced. Irrigation was applied when *c.* 50% of the water had evaporated (Fig. 6.1). The latter was considered to be the recommended level of depletion for vineyards to obtain a balance between yield and wine quality.

After one simulated irrigation season, simulated winter rainfall was applied to all treatments. Since the pot experiment was carried out in summer, there was no source of uncontaminated rainwater available. It was decided not to use distilled or de-ionized water for the rainfall simulations, since using distilled water changes the ionic balances and may flocculate or disperse the clay in the soil (Amezketa *et al*., 2004). Soil water contains solutes which are in balance with ions on exchange sites of the clay. Based on the forgoing, “rainwater” for the study was also abstracted from the Holsloot River, *i.e.* from a natural source where contamination is least expected (Table 6.2). The amount of rainfall applied to each soil was based on the long term mean rainfall for each of the different regions where the soil was collected (Appendix 6.1). During, and after each irrigation, as well as during and after each simulated rainfall day, the leachate was collected and pooled. The total volume of leachate per each soil was recorded at the end of the simulated rainfall period. The chemical status of the leachate from each soil was determined in samples collected from the pooled leachate.

6.2.5. Water sampling and analyses

Water samples were collected from the river water and wastewater tanks prior to each irrigation. The pH, electrical conductivity (EC), Na^+, K^+, Ca^{2+}, Mg^{2+}, Fe, Cl^-, HCO_3^-, SO_4^{2-}, B^- and COD in the water were determined at a commercial laboratory (BEMLAB, Strand). Details of the analytical procedures are described in Chapter 4.

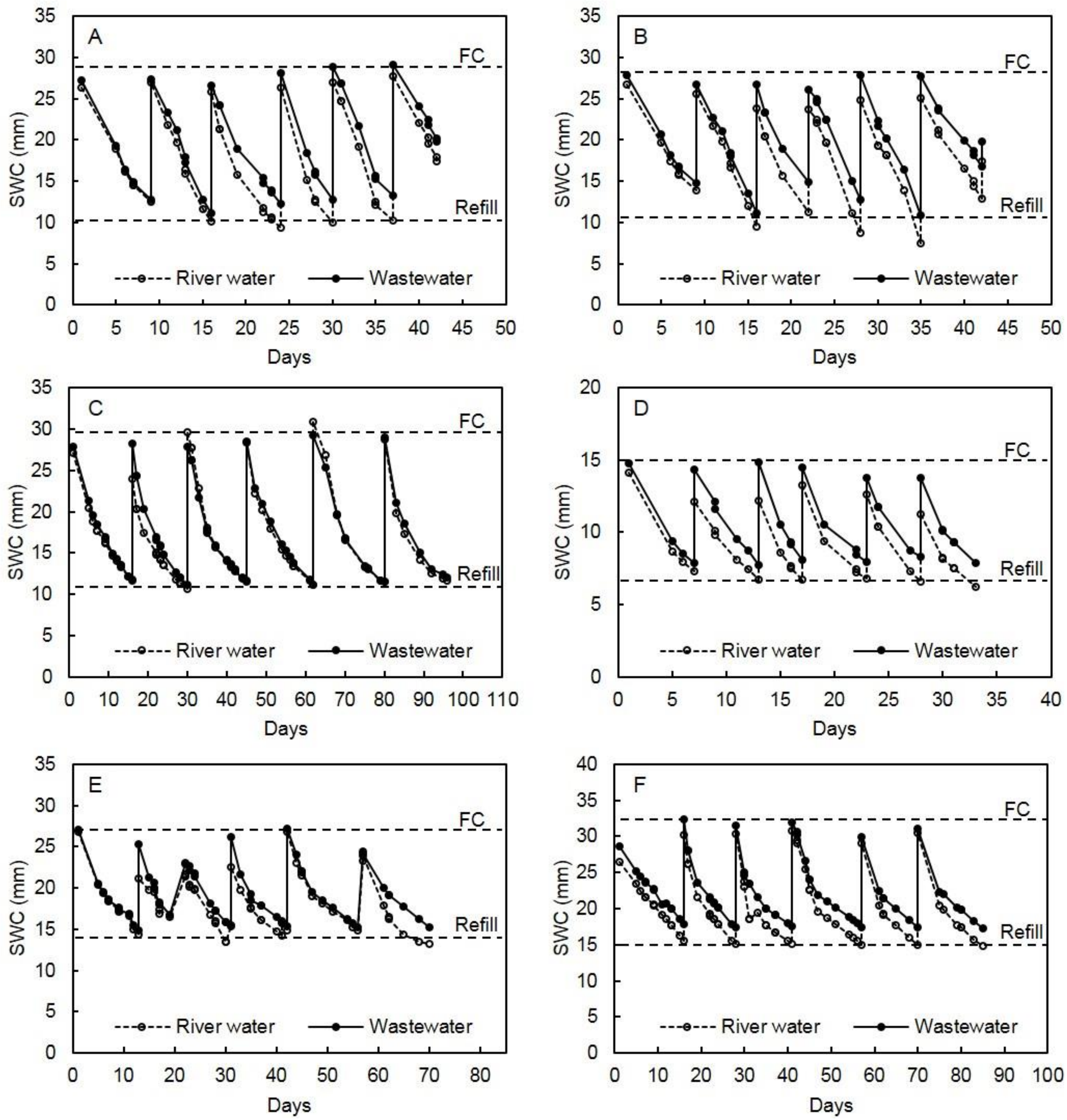

Figure 6.1. Variation in soil water content (SWC) in (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay soils where river water and diluted winery wastewater were applied to simulate one season's irrigation. The first irrigations were applied on 02-01-2013. Dashed horizontal lines indicate field capacity (FC) and the refill point.

Table 6.2. Chemical composition of raw water abstracted from the Holsloot River near Rawsonville used for simulating winter rainfall and water obtained from the Stellenbosch municipality.

Variable	Raw water	Municipal water
pH	6.1	7.2
EC (mS/m)	18.0	11.2
Na^{+} (mg/L)	10.0	9.3
K^{+} (mg/L)	1.1	1.4
Ca^{2+} (mg/L)	7.4	8.3
Mg^{2+}(mg/L)	3.5	2.1
SAR	0.76	0.75
Fe^{2+}(mg/L)	0.04	0.1
B^{3+} (mg/L)	0.03	0.01
Mn^{2+} (mg/L)	0.04	0.01
Zn^{2+} (mg/L)	0.83	0.19
P (mg/L)	0.01	0.04
NH_4^{+}-N (mg/L)	2.47	0.59
NO_3^{-} -N(mg/L)	0.95	0.11
Cl^{-} (mg/L)	89.8	21.2
HCO_3^{-} (mg/L)	15.3	26.7
SO_4^{2-}(mg/L)	6.1	3.2
TDS (mg/L)	75.1	45
COD (mg/L)	56	47

6.2.6. Soil sampling and analyses

After six irrigations, *i.e.* one simulated winery wastewater irrigation season, soil samples were collected from the 0 to 19 cm deep soil layer. Soil sampling was again carried out after the simulated winter rainfall had been applied. All analyses were carried out at a commercial laboratory (BEMLAB, Strand). Details of the analytical procedures are described in Chapter 4.

6.2.7. Soil characterization

The six soils selected, represent soils dominant in three of the main South African wine producing regions. It was expected that the effect of winter rainfall after winery wastewater irrigation would differ between different soils. The chemical properties of the six soils used, are presented in Table 6.3. The pH of the six soils used ranged between 4.2 and 6.6, whereas the EC_e ranged between 20 and 70 mS/m. The cation exchange capacity (CEC) ranged between 2.9 and 8.3 $cmol_c.kg^{-1}$ (Table 6.3). The Robertson soil had the highest clay content, *i.e*. 35%, whereas the Lutzville sand contained only 0.4% clay (Table 6.4).

Table 6.3. Chemical properties of the six soils before the river water and diluted winery wastewater irrigations were applied.

Variable	Rawsonville sand	Lutzville sand	Stellenbosch shale	Stellenbosch granite	Stellenbosch sand	Robertson clay
pH	5.8	6.5	4.2	4.3	4.3	6.6
EC_e (mS/m)	30	20	30	40	20	70
Bray2 P (mg/kg)	227	21	8	11	28	102
Bray2 K (mg/kg)	66	240	95	99	206	702
Org C (%)	0.8	0.2	1.5	1.3	0.9	0.6
K^{+}_{extr} ($cmol_c.kg^{-1}$)	0.17	0.61	0.24	0.25	0.53	1.79
Na^{+}_{extr} ($cmol_c.kg^{-1}$)	0.01	0.05	0.06	0.05	0.02	0.37
Ca^{2+}_{extr} ($cmol_c.kg^{-1}$)	2.79	3.22	1.64	1.21	0.7	9.17
Mg^{2+}_{extr} ($cmol_c.kg^{-1}$)	0.95	0.8	0.61	0.51	0.22	3.02
CEC ($cmol_c.kg^{-1}$)	3.9	3.4	4.3	3.6	2.9	8.3

Table 6.4. Particle size distribution and textural class of the six soils used in the study.

Particle size & textural class	Rawsonville sand	Lutzville sand	Stellenbosch shale	Stellenbosch granite	Stellenbosch sand	Robertson Clay
Clay *(<0.002 mm)*	3.3	0.4	20	13	7	35
Silt *(0.002-0.02 mm)*	1	1	13	17	6	20
Fine sand *(0.02-0.2 mm)*	60	69	50	33	39	35
Medium sand *(0.2-0.5 mm)*	29	26	5	3	26	7
Coarse sand *(0.5-2 mm)*	8	2	12	35	22	3
Textural class	Sand	Sand	Sandy clay loam	Sandy loam	Sand	Clay loam

6.2.8. Composition and amount of simulated winter rainfall applied

The overall average chemical composition of the Holsloot river water used to simulate winter rainfall was within the acceptable range for irrigation water (Table 6.2). The pH levels were below the recommended pH for irrigation water ranging from 6.5 to 8.4 (DWAF, 1996). The amount of rainfall applied was calculated from the long term average rainfall in the region where each soil was collected (Table 6.5 and Appendix 6.1). Rawsonville soils received the highest amount of rainfall per day (13.8 mm) followed by the three soils from Stellenbosch (9.3 mm), while the Robertson and Lutzville soils received the least rainfall (4.5 and 3.8 mm, respectively) (Table 6.5).

Table 6.5. Mean number of rainfall days, interval between rainfall days and amount of water per rainfall day during winter, *i.e.* from May until September, at the four localities where the soils were sampled, as well as the volume of water applied per pot to simulate the rainfall.

Soils	Number of rainfall days	Interval between rainfall days (days)	Amount per rainfall day	
			(mm/day)	(mL/pot)
Rawsonville sand	41	4	13.8	244
Lutzville sand	25	6	3.7	67
Stellenbosch shale	50	3	9.3	164
Stellenbosch granite	50	3	9.3	164
Stellenbosch sand	50	3	9.3	164
Robertson clay	34	5	4.5	80

6.2.9. Statistical procedures

The study was carried out under a 20 m x 40 m translucent fiberglass rain shelter at the ARC Infruitec-Nietvoorbij near Stellenbosch. Each soil/water treatment was replicated three times in a complete randomized design, *i.e.* 6 (soil) x 2 (water) x 3 (replicates). The six soils were randomly allocated within each block. The treatment design was a split-plot. The main plot factor was soil type and the sub-plot factor was soil depth. Analyses of variance were performed separately for each season using SAS version 9.2 (SAS, 2008). The Shapiro-Wilk test was performed to test for non-normality (Shapiro & Wilk, 1965). Student's t-least significant difference (LSD) was calculated at the 5% significance level to facilitate comparison between treatment means (Ott, 1998). STATGRAPHICS® was used to calculate the multiple linear regression equations.

6.3. RESULTS AND DISCUSSION

6.3.1. Chemical composition of the irrigation waters

The chemical composition of the river water quality was within the acceptable range for irrigation water (Table 6.6). The average water pH for six irrigations was 7.2 which is lower than the 8.4 which is the maximum threshold for irrigation (DWAF, 1996). The average EC value was 21 mS/m which was well below the 75 mS/m salinity threshold value for grapevine irrigation (Myburgh, 2012a). The average COD was 44.8 mg/L which in line with normal drinking water. The overall Na^+ and K^+ levels were very low (Table 6.6). With the exception of pH, winery wastewater chemical parameters were higher than those of the river water (Table 6.7). The most noticeable elements and properties that were higher in the wastewater were K^+, bicarbonate, EC, TDS and COD (Table 6.7). Although the average bicarbonate winery wastewater was high, it was high only in the first three irrigations while the last three irrigations it had dropped to almost zero. This could be attributed to the winery using different cleaning detergents during the latter period.

6.3.2. Comparison of the chemical status of the river water and actual rainfall

The river water used in the study contained substantially more basic cations compared to rainwater collected at Citrusdal and Cape Town (Fig. 6.2). In contrast, rainwater harvested at Kleinmond tended to contain more Na^+, K^+, and Ca^{2+} than the water abstracted from the Holsloot River. This suggested that the rainwater at Kleinmond was probably contaminated in the harvesting process. The cations in the river water was comparable to the water obtained from the Stellenbosch municipality (Fig. 6.2). It must be noted the level of Na^+ were higher compared to the other cations in all the waters. The pH in the river and municipal water tended to be slightly higher than in the rainwater (Fig. 6.3). The higher levels of cations caused the EC in the river and municipal water to be higher compared to the rainwater. The SAR in all the waters were relatively low, *i.e.* less than 1.5 (Fig. 6.3).

Table 6.6. Variation in quality of river water used for irrigation of six different soils in a pot experiment during a simulated winery wastewater irrigation season.

Water quality variable	Sampling date						
	03-01-2013	11-01-2013	18-01-2013	24-01-2013	06-02-2013	13-02-2013	Average
pH	7.6	7.8	7.6	6.8	7	6.2	7.2
EC (mS/m)	44	17	9	18.7	18.9	18.1	21.0
Na^{+} (mg/L)	53.9	11.6	12.6	12.9	42.4	13.5	24.5
K^{+} (mg/L)	5.6	3.4	2.6	2.7	3.1	2.5	3.3
Ca^{2+} (mg/L)	21.2	14.2	11.9	11.5	10.3	8.9	13.0
Mg^{2+} (mg/L)	11.5	5	5	5.2	8.3	5.5	6.8
SAR	2.4	0.7	0.8	0.8	2.4	0.9	1.3
Fe^{2+} (mg/L)	2.3	0.05	0.01	0.01	0	0	0.4
B^{3+} (mg/L)	0.01	0.02	0	0	0	0.02	0.0
Mn^{2+} (mg/L)	0.04	0.08	0.09	0.04	0.05	0.05	0.1
Cu^{2+} (mg/L)	0.03	0.02	0.01	0	0.01	0	0.0
Zn^{2+} (mg/L)	0.4	0.02	0.07	0.4	0.53	0.7	0.4
P (mg/L)	0.2	0.12	0	0.02	0	0.03	0.1
NH_4^{+}-N (mg/L)	0.5	0.4	0.4	1.9	0.23	2.04	0.9
NO_3^{-}-N (mg/L)	14	0.5	1.1	0	2.7	0.8	3.2
Cl^{-} (mg/L)	72.4	27.7	28.9	31	22.1	39.8	37.0
HCO_3^{-} (mg/L)	55.1	10.4	12.1	9.2	15.3	22.9	20.8
SO_4^{2-} (mg/L)	27.4	25.4	24.5	54	32	33	32.7
TDS (mg/L)	264	45	52	119	121	116	120
COD (mg/L)	48	62	57	18	38	46	45

Table 6.7. Quality variation of winery wastewater diluted to 3000 mg/L COD for irrigation of six different soils in a pot experiment during a simulated winery wastewater irrigation season.

Water quality variable	Sampling date						
	03-01-2013	11-01-2013	18-01-2013	24-01-2013	06-02-2013	13-02-2013	Average
pH	6.1	6.9	7.9	4.9	4.1	3.9	5.6
EC (mS/m)	236	212	246	39	45	41	137
Na^+ (mg/L)	65	69	86	20	26	19	48
K^+ (mg/L)	487	422	663	55	50	45	287
Ca^{2+} (mg/L)	45.1	70.3	43.2	21.2	13.7	14.9	34.7
Mg^{2+} (mg/L)	39.1	44.6	62.4	10.1	9.5	9.5	29.2
SAR	1.7	1.6	2.0	0.9	1.3	7.1	2.4
Fe^{2+} (mg/L)	4.7	3.5	1.5	1.9	4.8	4.01	3.4
B^{3+} (mg/L)	0.44	0.6	0.8	0.11	0	0.14	0.3
Mn^{2+} (mg/L)	0.4	0.7	0.5	0.2	0.2	0.16	0.4
Cu^{2+} (mg/L)	0.03	0.04	0.2	0.1	0.1	0.23	0.1
Zn^{2+} (mg/L)	0.02	0.11	0.5	0.5	0.9	1.39	0.6
P (mg/L)	21.9	26.4	42.5	6	6.1	6.12	53
NH_4^+-N (mg/L)	12	8.5	17.2	2.2	0.8	1	687
NO_3^--N (mg/L)	240	4.9	0.3	0	0.8	1.34	187
Cl^- (mg/L)	57	82	51	34	45	48	0.3
HCO_3^- (mg/L)	975	1047	2102	0.01	0.1	0.2	0.4
SO_4^{2-} (mg/L)	765	39	123	43	117	36	0.1
TDS (mg/L)	1418	567	1490	246	289	245	709
COD (mg/L)	3080	2870	3460	3540	3350	3500	3300

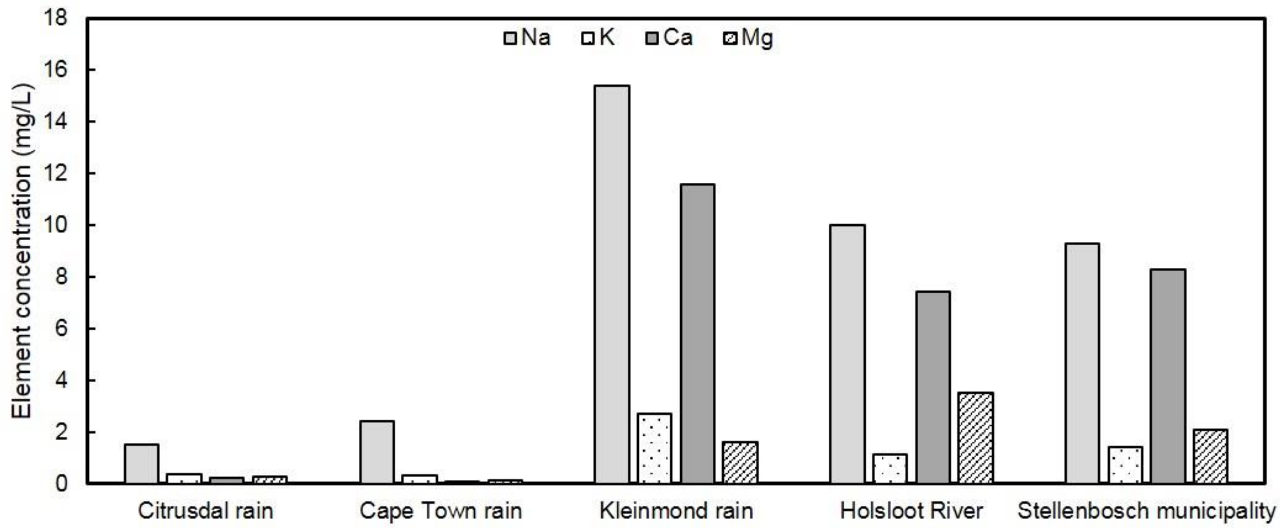

Figure 6.2. Basic cation concentration in rainwater collected at Citrusdal and Cape Town (after Soderberg, 2003) and rainwater harvested at Kleinmond (after Dobrowsky, 2014) compared to water obtained from the Holsloot River near Rawsonville and the Stellenbosch municipality, respectively.

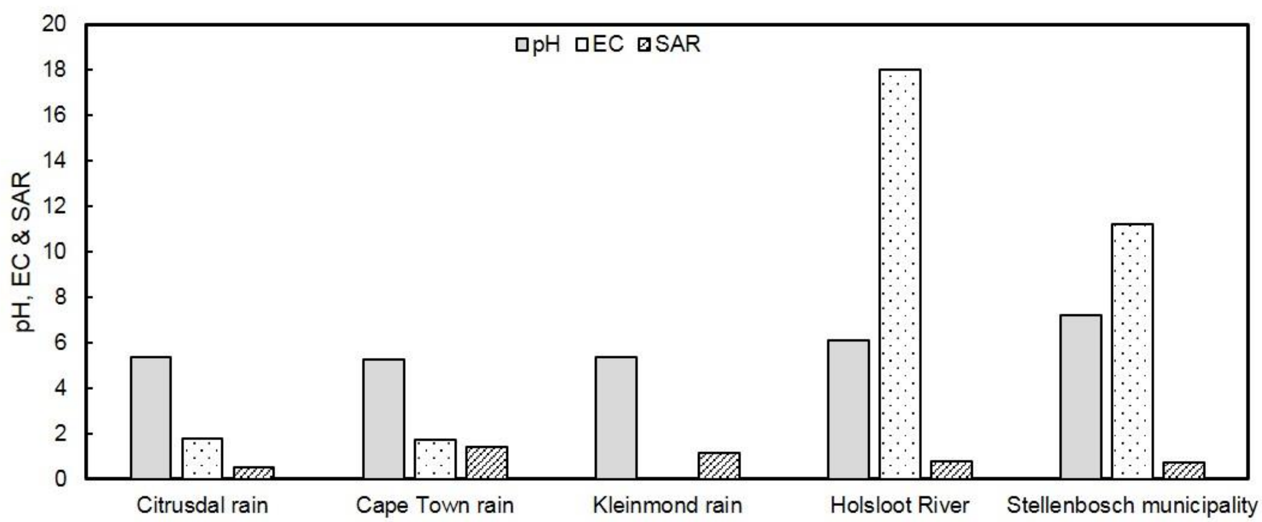

Figure 6.3. The pH, electrical conductivity (EC) and sodium adsorption ratio (SAR) in rainwater collected at Citrusdal and Cape Town (Soderberg, 2003) and rainwater harvested at Kleinmond (Dobrowski, 2014) compared to water obtained from the Holsloot River near Rawsonville and the Stellenbosch municipality, respectively. The EC was not determined at Kleinmond.

6.3.3. Composition of leachate after simulated winter rainfall

No leachate occurred in the case of the Lutzville and Robertson soils after the simulated winter rainfall had been applied. This indicated that the volumes of simulated rainfall was inadequate to leach the elements applied *via* wastewater nutrients from these soils. In the case of the Robertson clay, a high water holding capacity also could have prevented leaching of solutes. For the soils where leaching occurred, extremely small volumes of solutes leached following a simulated rainfall event. Only 0.83±0.15, 0.36±0.14, 0.55±0.16 and 0.64±0.14 mL were collected per rainfall event for the Rawsonville sand, Stellnbosch shale, Stellenbosch granite and Stellenbosch sand, respectively. The total leachate, *i.e.* which was used for the chemical analyses amounted respectively to 34.5, 18.6, 27.9 and 32.7 mL per treatment replication for the four soils. It must be noted that the same rainfall was applied to the three Stellenbosch soils.

The chemical composition of the leachates varied considerably (Table 6.8). In some cases, the pH, EC and element concentrations were unexpectedly higher in leachates where river water was used for irrigation before the simulated rainfall compared to irrigation with winery wastewater. It should be noted that the K^+ and COD in the leachate from the Stellenbosch sand was substantially higher compared to the other soils, particularly where winery wastewater was applied (Table 6.8).

Differences in the composition of the leachates were clearly reflected in their different colours observed (Fig. 6.4). Winter rainfall following winery wastewater irrigation caused leaching of K^+ and Na^+ from the soil as explained earlier. The leachate from soils irrigated with winery wastewater, were darker in colour than those irrigated with river water indicating leaching of organic matter. In the case of the Stellenbosch sand, the color of the leachate was similar to that of the winery wastewater applied (Fig. 6.4D). This indicated that substantial leaching of organic compounds applied *via* the winery wastewater occurred compared to the other soils.

Table 6.8. Chemical composition of the leachate collected after simulated winter rainfall for soils that were first irrigated with river water (RW) and winery wastewater (WW), respectively. No leachate could be obtained for the Lutzville and Robertson soils.

Water quality variable	Rawsonville sand		Stellenbosch shale		Stellenbosch granite		Stellenbosch sand	
	RW	WW	RW	WW	RW	WW	RW	WW
pH	7.1	7.5	6.7	6.5	6.3	6.5	6.1	7.2
EC (mS/m)	133.5	84.4	21.5	28.8	18	33.1	97.8	86.6
Na^{+} (mg/L)	58.8	53	17.2	25.8	15.9	31	54.3	73.5
K^{+} (mg/L)	16.1	81.3	6	24.6	7.6	52.4	147.7	242.3
Ca^{2+} (mg/L)	132.9	57.6	19.4	18.9	14.8	12.4	69.9	28.5
Mg^{2+} (mg/L)	87.3	33.9	7.2	7.5	5.8	6.3	23.8	11.8
SAR (mg/L)	0.06	0.06	0.17	0.17	0.2	0.2	0.92	0.25
Fe^{2+} (mg/L)	0.04	0.07	2.39	11.06	2.2	2.02	0.22	12.21
P (mg/L)	0.04	0.08	0.02	0.04	0.01	0.04	0.07	0.19
NO_3^{-}-N	0	0	0.07	0.04	0.05	0	0.15	0.08
Cl^{-} (mg/L)	185.8	106.2	31	53.1	35.4	57.5	92.9	119.4
HCO_3^{-} (mg/L)	53.6	145.5	15.3	7.7	7.7	23	7.7	183.7
SO_4^{2-} (mg/L)	95	65	17	32	18	35	61	82
TDS(mg/L)	0.04	0.06	0.02	0.03	0.02	0.03	0.03	0.08
COD	0.94	1.1	0.09	0.17	0.13	0.32	0.18	2.1

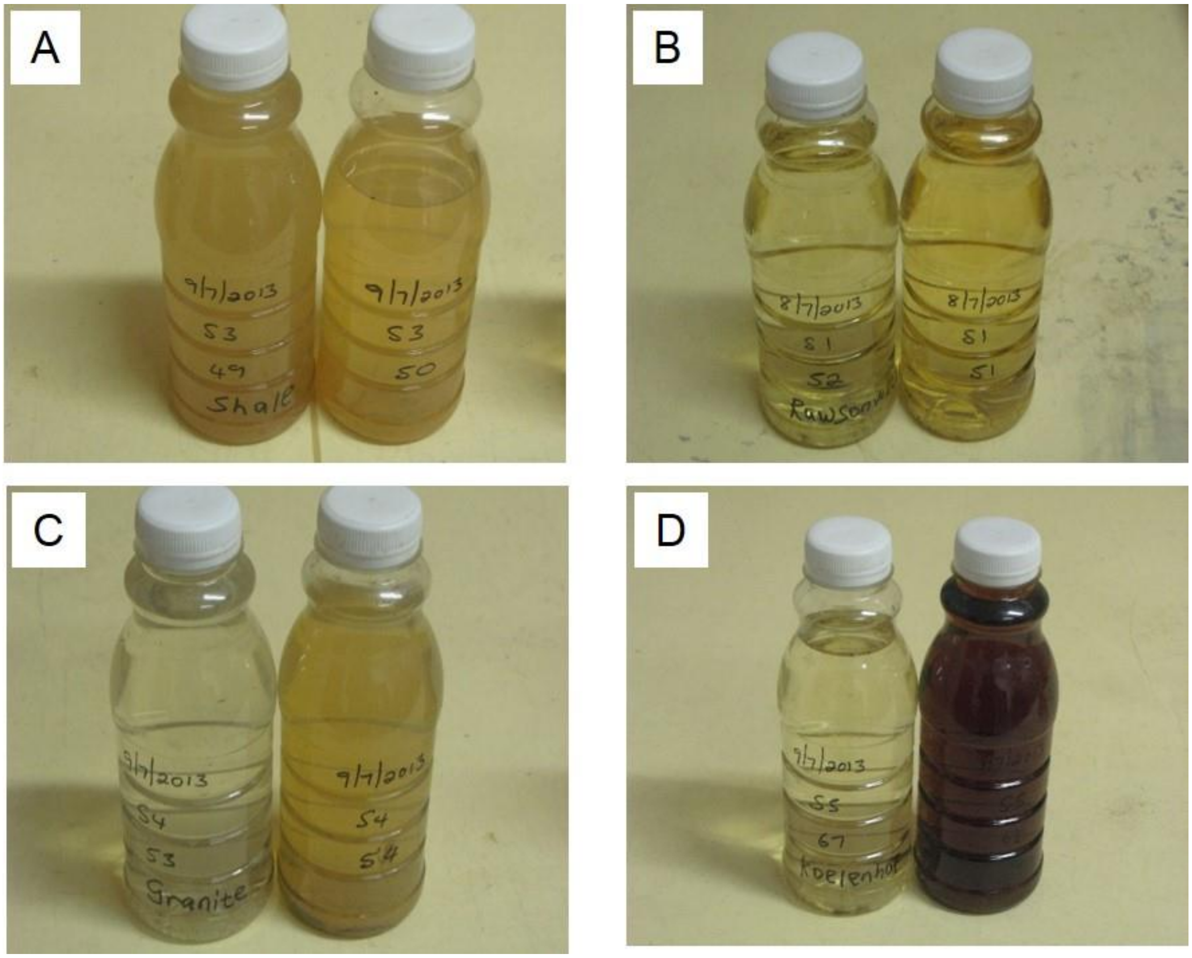

Fig 6.4. Examples of leachate collected after simulated winter rainfall from (A) Rawsonville sand, (B) Stellenbosch shale, (C) Stellenbosch granite and (D) Stellenbosch sand. Bottles on the left contains leachate where river water was applied.

6.3.4. Amount of cations leached

The amount of cations leached from the four soils where percolation occurred, were relatively low both where river water and diluted winery wastewater were applied. With the exception of K^{+}_{extr}, the amount of cations in the leachate declined non-linearly as the clay content increased (Fig. 6.5). Where river water was used for irrigation, the leached K^{+}_{extr} was relatively high for the Stellenbosch sand, and low for the Rawsonville sand (Fig. 6.5A). In the case of the Stellenbosch sand, substantially more K^{+}_{extr} also leached where the wastewater was applied compared to the other soils (Fig. 6.5B). Perusal of the data revealed that the K^{+}_{extr} variation seemed to be a function of (i) the initial K^{+}_{extr} in the soil before the irrigations were applied ($K_{initial}$) and (ii) the level of COD in the leachate

($COD_{leachate}$). Based on this, c. 90% of the variation in leached K^+ could be explained by means of the following multiple linear regression equation.

$K_{leached} = 7.262*K_{initial} + 1.145*COD_{leachate} - 1.58$ (R^2= 0.9279; s.e. = 0.42; p = 0.0006) (Eq 6.1)

The high K^+_{extr} content in the Stellenbosch sand compared to the other soils, probably contributed to the high level in the leachate. Since the COD was higher in the Stellenbosch sand, it suggested that the K^+ formed organic salts which readily leached from the soil, particularly where the winery wastewater was applied prior to the simulated rainfall. These organic compounds probably contributed to the dark colour of the leachate from the Stellenbosch sand (Fig. 6.4D). Surprisingly, the CEC did not make a significant contribution to the multiple linear regression model. However, this does not rule out the possibility that the relatively low CEC of the Stellenbosch sand (Table 6.3) could have played a minor role.

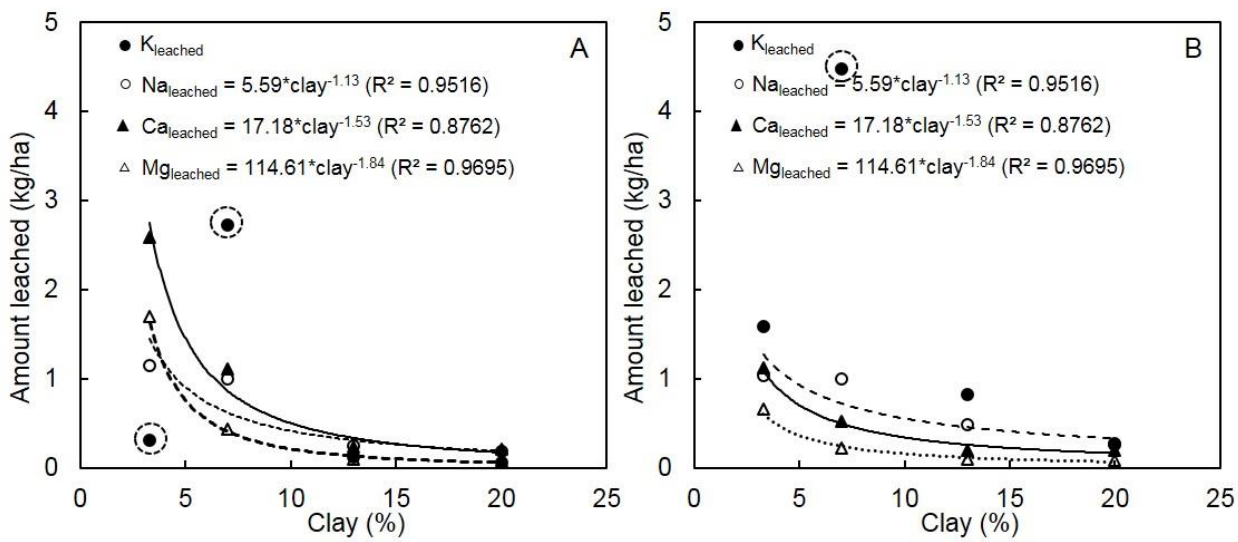

Figure 6.5. Relationship between amount of basic cation leached and the clay content where simulated winter rainfall was applied to soils that were first irrigated with (A) river water and (B) diluted winery wastewater, respectively. Due to the encircled outliers, K^+ could not be related to the clay content.

6.3.5. Calculated cation balances

Where river water was applied, the fraction of applied basic cations leached from the six soils tended to be higher compared to irrigation diluted with winery wastewater (Tables 6.9 to 6.12).

Table 6.9. Balance of K^+ applied *via* simulated irrigation and winter rainfall where six soils were irrigated with river water and winery wastewater, respectively.

Soils	Applied (kg/ha)			Retained (kg/ha)	Leached (kg/ha)	Retained (%)	Leached (%)
	Via irrigation	*Via* rainfall	Total				
				River water			
Rawsonville sand	6.0	6.2	12.2	11.9	0.31	97.4	2.57
Lutzville sand	6.7	1.0	7.7	7.7	0.00	100.0	0.00
Stellenbosch shale	4.4	5.1	9.5	9.5	0.06	99.3	0.66
Stellenbosch granite	3.5	5.1	8.6	8.5	0.12	98.6	1.39
Stellenbosch sand	5.0	5.1	10.1	7.4	2.73	73.0	26.97
Robertson clay	5.0	1.7	6.7	6.7	0.00	100.0	0.00
				Diluted winery wastewater			
Rawsonville sand	2772.3	6.2	2778.5	2776.9	1.58	99.9	0.06
Lutzville sand	2699.2	1.0	2700.2	2700.2	0.00	100.0	0.00
Stellenbosch shale	2673.9	5.1	2679.0	2678.7	0.26	100.0	0.01
Stellenbosch granite	1874.2	5.1	1879.3	1878.4	0.83	100.0	0.04
Stellenbosch sand	2717.7	5.1	2722.8	2718.4	4.48	99.8	0.16
Robertson clay	2276.5	1.7	2278.1	2278.1	0.00	100.0	0.00

Table 6.10. Balance of Na^+ applied *via* simulated irrigation and winter rainfall where six soils were irrigated with river water and winery wastewater, respectively.

Soils	Applied (kg/ha)			Retained (kg/ha)	Leached (kg/ha)	Retained (%)	Leached (%)
	via irrigation	*via* rainfall	Total				
River water							
Rawsonville sand	44.5	56.6	101.1	99.9	1.15	98.9	1.13
Lutzville sand	50.0	9.3	59.3	59.3	0.00	100.0	0.00
Stellenbosch shale	32.8	46.5	79.3	79.1	0.18	99.8	0.23
Stellenbosch granite	25.9	46.5	72.4	72.1	0.25	99.7	0.35
Stellenbosch sand	34.0	46.5	80.5	79.5	1.00	98.8	1.25
Robertson clay	39.0	15.3	54.3	54.3	0.00	100.0	0.00
Diluted winery wastewater							
Rawsonville sand	458.8	56.6	515.4	514.4	1.03	99.8	0.20
Lutzville sand	446.7	9.3	456.0	456.0	0.00	100.0	0.00
Stellenbosch shale	442.5	46.5	489.0	488.8	0.27	99.9	0.06
Stellenbosch granite	310.2	46.5	356.7	356.2	0.49	99.9	0.14
Stellenbosch sand	449.8	46.5	496.3	495.3	1.00	99.8	0.20
Robertson clay	376.8	15.3	392.1	392.1	0.00	100.0	0.00

Table 6.11. Balance of Ca^{2+} applied *via* simulated irrigation and winter rainfall where six soils were irrigated with river water and winery wastewater, respectively.

Soils	Applied (kg/ha)			Retained (kg/ha)	Leached (kg/ha)	Retained (%)	Leached (%)
	via irrigation	*via* rainfall	Total				
				River water			
Rawsonville sand	23.6	41.9	65.5	62.9	2.59	96.0	3.96
Lutzville sand	26.5	6.8	33.3	33.3	0.00	100.0	0.00
Stellenbosch shale	17.4	34.4	51.8	51.6	0.20	99.6	0.39
Stellenbosch granite	13.7	34.4	48.1	47.9	0.23	99.5	0.49
Stellenbosch sand	18.0	34.4	52.4	51.3	1.11	97.9	2.11
Robertson clay	21.0	11.3	32.3	32.3	0.00	100.0	0.00
				Diluted winery wastewater			
Rawsonville sand	335.5	41.9	377.4	376.3	1.12	99.7	0.30
Lutzville sand	326.7	6.8	333.5	333.5	0.00	100.0	0.00
Stellenbosch shale	323.6	34.4	358.0	357.8	0.20	99.9	0.06
Stellenbosch granite	226.8	34.4	261.2	261.0	0.20	99.9	0.07
Stellenbosch sand	328.9	34.4	363.3	362.8	0.53	99.9	0.14
Robertson clay	275.5	11.3	286.8	286.8	0.00	100.0	0.00

Table 6.12. Balance of Mg^{2+} applied *via* simulated irrigation and winter rainfall where six soils were irrigated with river water and winery wastewater, respectively.

Soils	Applied (kg/ha)			Retained (kg/ha)	Leached (kg/ha)	Retained (%)	Leached (%)
	via irrigation	*via* rainfall	Total				
				River water			
Rawsonville sand	12.3	19.8	32.1	30.4	1.70	94.7	5.30
Lutzville sand	13.9	3.2	17.1	17.1	0.00	100.0	0.00
Stellenbosch shale	9.1	16.3	25.4	25.3	0.08	99.7	0.30
Stellenbosch granite	7.2	16.3	23.5	23.4	0.09	99.6	0.39
Stellenbosch sand	9.0	16.3	25.3	24.8	0.44	98.3	1.74
Robertson clay	11.0	5.4	16.4	16.4	0.00	100.0	0.00
				Diluted winery wastewater			
Rawsonville sand	282.1	19.8	301.9	301.2	0.66	99.8	0.22
Lutzville sand	274.6	3.2	277.9	277.9	0.00	100.0	0.00
Stellenbosch shale	272.0	16.3	288.3	288.2	0.08	100.0	0.03
Stellenbosch granite	190.7	16.3	207.0	206.9	0.10	100.0	0.05
Stellenbosch sand	276.5	16.3	292.8	292.6	0.22	99.9	0.07
Robertson clay	231.6	5.4	237.0	237.0	0.00	100.0	0.00

The fraction of applied cations leached also varied considerably between the different soil-rainfall combinations. Although the winter rainfall at Rawsonville was highest of the four localities, most of the applied cations were retained in this sandy soil. Due to the low winter rainfall in the Lower Olifants river region, no cations were leached from the Lutzville sand (Tables 6.9 to 6.12). The Stellenbosch shale and granite soils also retained most of the applied cations (Tables 6.9 to 6.12). The highest fraction of cations leached from the Stellenbosch sand. Due to the relatively low rainfall in the Breede River valley and a high water holding capacity, all the applied cations were retained by the Robertson soil (Tables 6.9 to 6.12).

The amounts of basic cations leached from the Rawsonville sand and the Stellenbosch sand reported for the field study in Chapter 3 were considerably higher than the amounts that leached in the pot experiment (Tables 6.9 to 6.12). This suggested that the rainfall did not play a prominent role under field conditions. In fact, the rainfall at Rawsonville and Stellenbosch were substantially lower than the large volumes of irrigation applied to the grazing paddocks in the field study. Therefore, it seems that the irrigation was responsible for the leaching of cations in the field, and it is not an environment friendly way of wastewater disposal. If diluted winery wastewater is to be used for vineyard irrigation, the irrigation volumes will be relatively low. This suggests that little, or no leaching, might occur under normal rainfall conditions, and that cations will accumulate in the soil. Since rainfall in South Africa is highly variable (Dent *et al.*, 1987), it does not rule out the possibility that abnormally high daily rainfall events can leach the cations from vineyard soils irrigated with winery wastewater. In fact, a field study showed that cations accumulated during summer where grapevines in a sandy soil near Rawsonville were irrigated with winery wastewater diluted to 3000 mg/L COD (Howell & Myburgh, 2014). However, the accumulated cations, particularly K^+ and Na^+, were leached out beyond 1.8 m depth during winter. Since rainfall events of up to 80 mm/d were recorded (Howell & Myburgh, 2014), it indicated that high rainfall events could leach accumulated cations where winery wastewater is used for irrigation. Furthermore, it is possible that occasional freak floods could leach accumulated cations from the soil.

In the case of the diluted winery wastewater irrigations, perusal of the data revealed that the variation in the amount of a specific cation retained in the soils appeared to be a function of (i) the element amount applied *via* the irrigation plus rainfall, (ii) the amount of rainfall and (iii) the organic carbon content of the soil. Using these three variables, most of the variation in the amount of a specific cation could be explained by means of multiple linear regression models for all the soils (Table 6.13). According to these models, the retention of cations increased with the amount of element applied. As expected, the retained amount decreased as the winter rainfall increased. The amount of element retained also increased with the organic carbon in the soil. The fact that the organic compunds can increase the CEC of soils, is well-documented (Harada & Inoko, 1975; Parfitt *et al.*, 1995; Caravaca, 1999; Seilsepour & Rashidi, 2008).

The equations in Table 6.13 were used to estimate the amount of cations that would have remained in the soil after the rainfall simulation if the water used contained no cations. Due to the relatively low level of K^+ in the water (Fig. 6.2), the difference between the actual and estimated amount of K^+ retained was relatively small, *i.e.* less than 0.5% (Table 6.14). However, in the case of Na^+ and Ca^{2+} the differences in the amounts retained were as high as 13%. In the case of Mg^{2+} the differences were less than 8%. This indicated that the higher levels of Na^+, Ca^{2+} and Mg^{2+} in the river water used, increased the actual amount of cations retained, irrespective of soil type. Therefore, harvesting rainwater for leaching studies would be advisable. However, care should be taken to avoid contamination of the harvested water, particularly if it needs to be stored in tanks.

Table 6.13. Slopes (m_n), constants, correlation coefficients (R^2), standard error (s.e.), level of significance (*P*) and number of data sets (n) used for multiple linear regression models to estimate the amount of cations retained in the soil after simulated irrigation with diluted winery wastewater was followed by simulated winter rainfall, with the amount applied, winter rainfall and organic carbon content as the independent variables.

Basic cation	Applied (kg/ha)	Rainfall (mm)	Organic C (%)	Constant	R^2	s.e.	*P*	n
	m_1	m_2	m_3					
K^+	0.9996	-0.0077	1.8589	1.1299	0.9999	2.07	0.0001	6
Na^+	0.9997	-0.0035	0.7964	0.2078	0.9998	0.41	0.0001	6
Ca^{2+}	0.9991	-0.0029	0.7851	0.3316	0.9998	0.15	0.0001	6
Mg^{2+}	0.9998	-0.0029	0.4865	0.0729	0.9998	0.12	0.0001	6

Table 6.14. Difference between actual amount of basic cation retained by six soils and the estimated amount that would have been retained if the water used for the rainfall simulation contained none of the four basic cations.

Cation	Soil	Amount of element retained (kg/ha)		Difference
		Actual	Estimated	(%)
K^{+}	Rawsonville sand	466.7	459.1	-0.43
	Lutzville sand	450.9	449.5	-0.15
	Stellenbosch shale	450.5	444.8	-0.32
	Stellenbosch granite	316.6	311.5	-0.42
	Stellenbosch sand	453.6	450.9	-0.36
	Robertson clay	381.1	379.5	-0.17
Na^{+}	Rawsonville sand	514.4	457.5	-11.05
	Lutzville sand	456.0	446.6	-2.05
	Stellenbosch shale	488.8	442.2	-9.54
	Stellenbosch granite	356.2	309.7	-13.06
	Stellenbosch sand	494.9	448.9	-9.29
	Robertson clay	392.1	376.8	-3.89
Ca^{2+}	Rawsonville sand	376.3	334.4	-11.12
	Lutzville sand	333.5	326.5	-2.11
	Stellenbosch shale	357.8	323.3	-9.64
	Stellenbosch granite	261.0	226.5	-13.24
	Stellenbosch sand	362.8	328.2	-9.54
	Robertson clay	286.8	275.5	-3.95
Mg^{2+}	Rawsonville sand	301.2	281.5	-6.54
	Lutzville sand	277.9	274.6	-1.18
	Stellenbosch shale	288.2	272.0	-5.63
	Stellenbosch granite	206.9	190.6	-7.88
	Stellenbosch sand	292.6	276.2	-5.60
	Robertson clay	237.0	231.7	-2.23

6.3.6. Soil chemical changes after irrigation and simulated winter rainfall

6.3.6.1. Basic cations

Potassium: Similar to the results reported in Chapter 4, the soil K^+_{extr} showed almost no change where river water was applied, irrespective of soil type (Fig. 6.6). In contrast, irrigation with winery wastewater increased the K^+_{extr} in the six soils. Due to no, or limited leaching, as discussed above, the K^+_{extr} remained almost unchanged in all soils after the simulated winter rainfall. In fact, K^+_{extr} in the Lutzville sand tended to increase slightly after the rainfall (Fig. 6.6B). The Stellenbosch sand was the only soil where the K^+_{extr} showed a prominent decline after the rainfall (Fig. 6.6E). This trend was probably due the combined effect of the high initial K^+_{extr} content and the organic compounds on the amount of K^+ leached from the soil as discussed above. In spite of the leaching, the soil K^+_{extr} retained was still almost double the initial level.

Sodium: The Na^+_{extr} tended to increase in all soils where river water was used for irrigation (Fig. 6.7). This due to the relatively high Na^+ content in the water (Table 6.6), *i.e.* on average 24.5 mg/L compared to 3.3 mg/L for K^+. As expected, irrigation with the diluted winery wastewater increase the Na^+_{extr} in all soils to higher levels than river water irrigation in most of the soils (Fig. 6.7). The levels of Na^+_{extr} in the Lutzville sand and Robertson clay were similar after the river water, as well as winery wastewater irrigations (Figs. 6.7B & 6.7F). At this stage, there is no explanation for this unexpected result, other than possible experimental errors. The Na^+_{extr} declined in all the soils where the simulated rainfall resulted in leaching (Fig. 6.7). Where river water was used for irrigation, the Na^+_{extr} levels were comparable to, or even lower in the Rawsonville sand, the initial levels. However, the simulated rainfall was insufficient to leach all the Na^+ where diluted winery wastewater was used for irrigation. Due to the relatively high Na^+ in the water used for the rainfall simulation (Fig. 6.2), the Na^+_{extr} in the Lutzville sand and Robertson clay increased after the rainfall (Figs. 6.7B & 6.7F). This indicated that the Na^+ in the river water used for the rainfall simulation contributed to the soil Na^+_{extr} where diluted winery wastewater was applied. This confirms the importance of using natural rainfall for leaching studies.

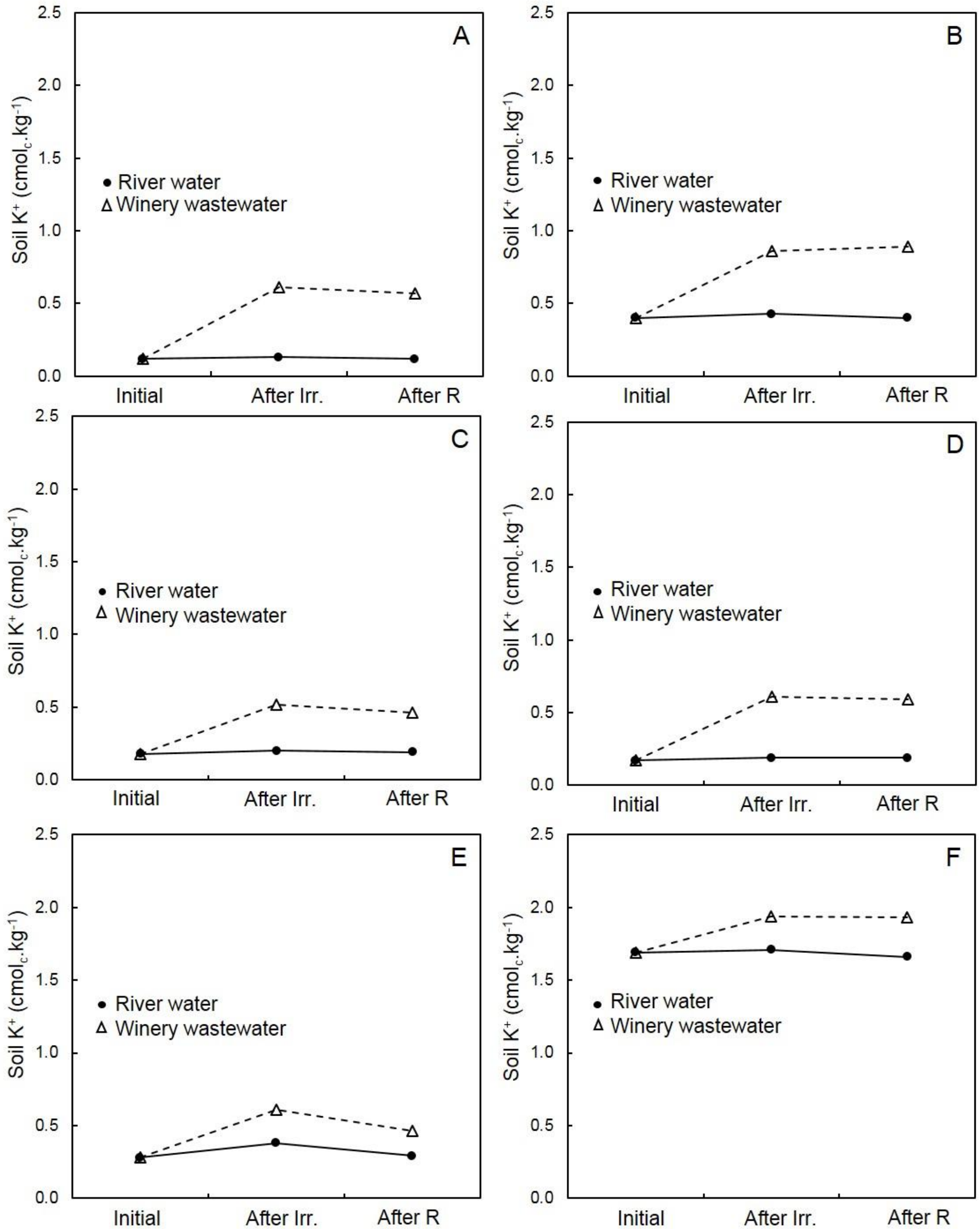

Figure 6.6. Effect of irrigation with river water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on extractable soil K^+ for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

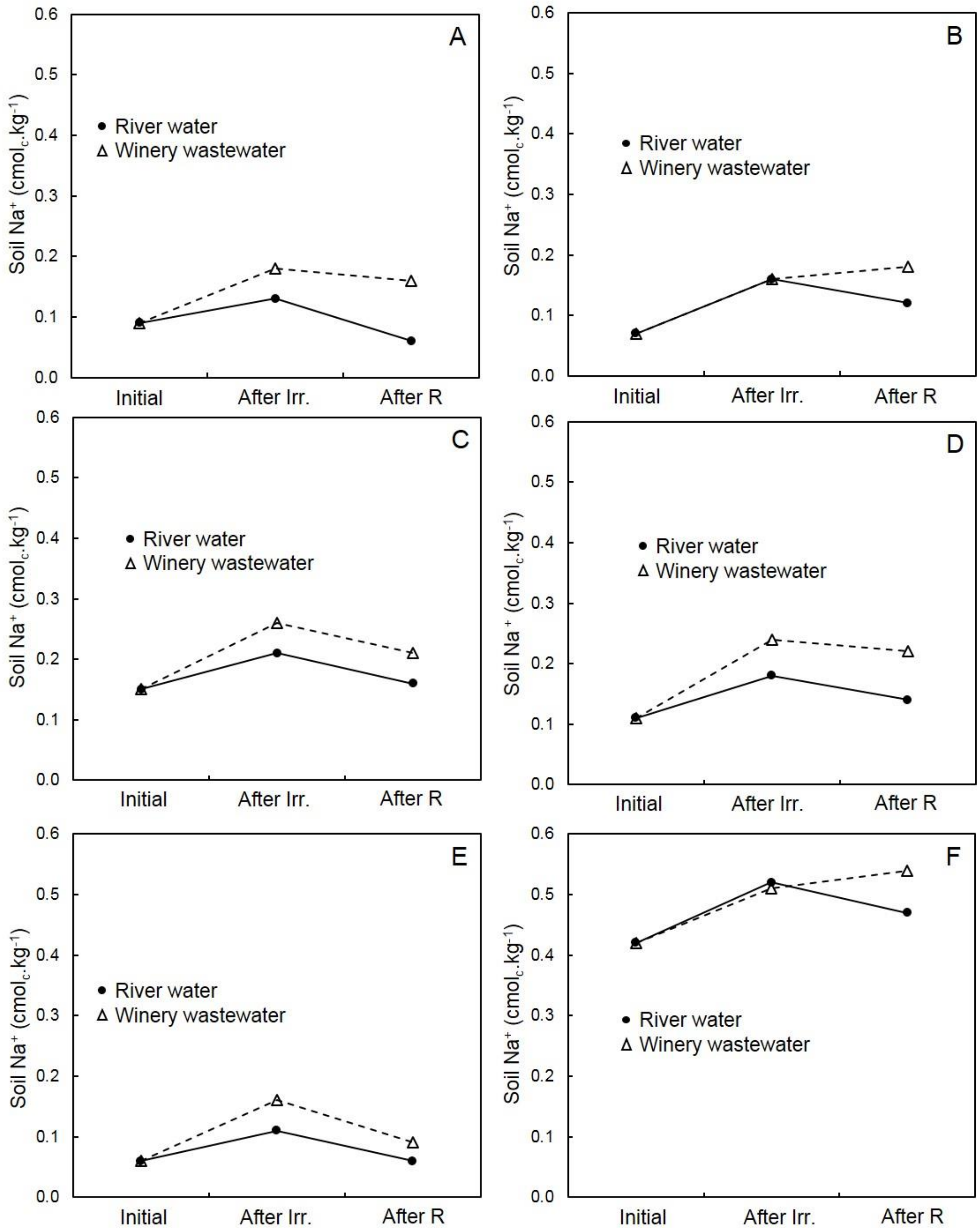

Figure 6.7. Effect of irrigation with river water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on extractable soil Na^+ for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

Calcium: Although more Ca^{2+} was applied *via* the diluted winery wastewater (Table 6. 11), it had no effect on the level of soil Ca^{2+}_{extr} compared to the irrigation with river water, irrespective of soil type (Fig. 6.8). This indicated that the *c.* 35 mg/L Ca^{2+} in the wastewater (Table 6.7) was too low to increase the soil Ca^{2+}_{extr}. Likewise, the simulated rainfall had no effect on the soil Ca^{2+}_{extr}, except for a substantial increase in the Robertson clay (Fig. 6.8F). At this stage, there is no explanation for this unexpected result.

Magnesium: Similar to Ca^{2+}_{extr}, irrigation with diluted winery wastewater had no effect on the level of soil Mg^{2+}_{extr} compared to the irrigation with river water, irrespective of soil type (Fig. 6.9). The simulated rainfall had no effect on the Mg^{2+}_{extr} in most soils. However, the Mg^{2+}_{extr} increased except Stellenbosch granite that was irrigated with winery wastewater before the rainfall simulation (Fig. 6.9D). The Mg^{2+}_{extr} in the Robertson clay did not increased after the simulated rainfall, irrespective of the water used for irrigation (Fig. 6.9E). At this stage, there is no explanation for these unexpected results.

6.3.6.2. Soil EPP' and ESP'

EPP': Irrigation with river water tended to decrease the EPP' in some of the soils (Fig. 6.10). The lower EPP' probably resulted from the increase in soil Na^{+}_{extr} (Fig. 6.7) caused by the relatively high Na^{+} compared to K^{+} in the river water as discussed above. Since the Ca^{2+}_{extr} (Fig. 6.8) and Mg^{2+}_{extr} remained constant (Fig. 6.9), it did not affect the EPP'. Irrigation with diluted winery wastewater increased the EPP' in all soils, except for the Robertson clay where the EPP' remained almost the same (Fig. 6.10). The higher EPP' resulted from the increase in soil K^{+}_{extr} (Fig. 6.6) whereas the Ca^{2+}_{extr} and Mg^{2+}_{extr} remained constant.

In the soils that were irrigated with river water, the simulated rainfall caused a further decline in EPP' in most soils (Fig. 6.10). This was also due to the relatively low K^{+} in comparison to the other cations in the river water used for the rainfall simulation. In soils that were irrigated with diluted winery wastewater, and where leaching occurred, the simulated rainfall caused a decline in EPP' (Fig. 6.10). However, in the case of the Lutzville sand and the Robertson clay where no leaching of K^{+} occurred, the EPP' also tended to decrease as a result of the low ratio of K^{+} versus the other cations in the river water (Figs. 6.10B and 6.10F).

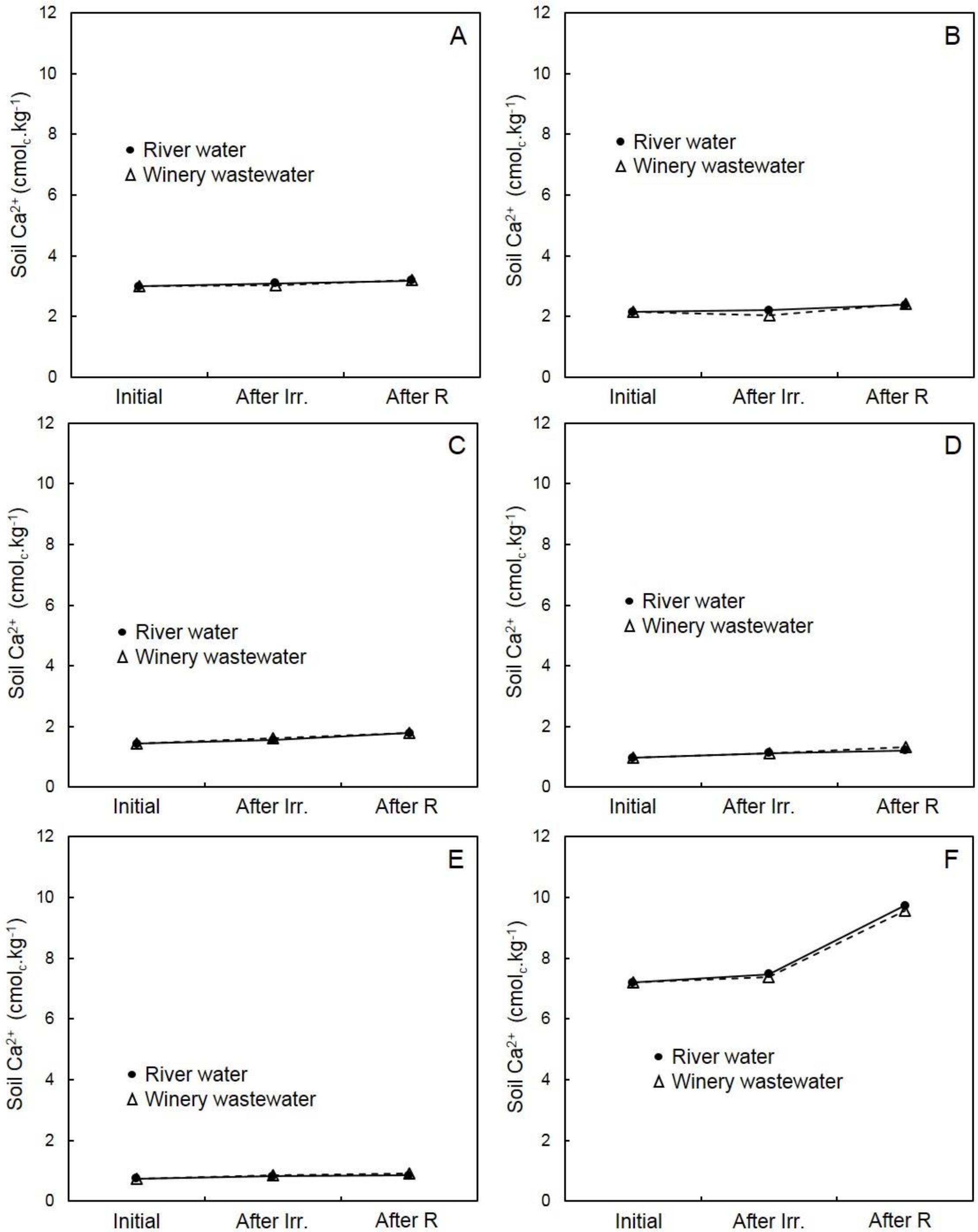

Figure 6.8. Effect of irrigation with clean water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on extractable soil Ca^{2+} for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

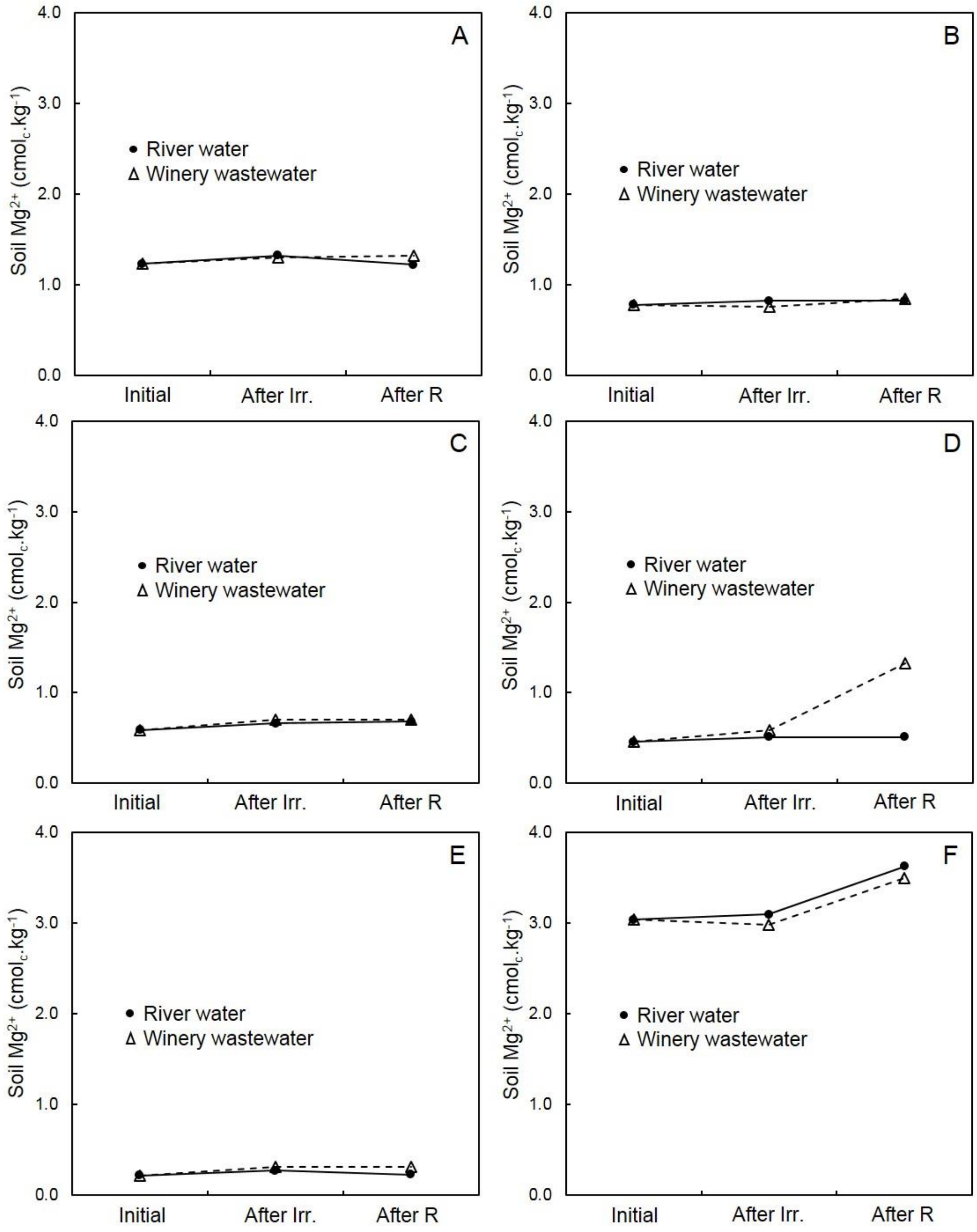

Figure 6.9. Effect of irrigation with clean water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on extractable soil Mg^{2+} for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

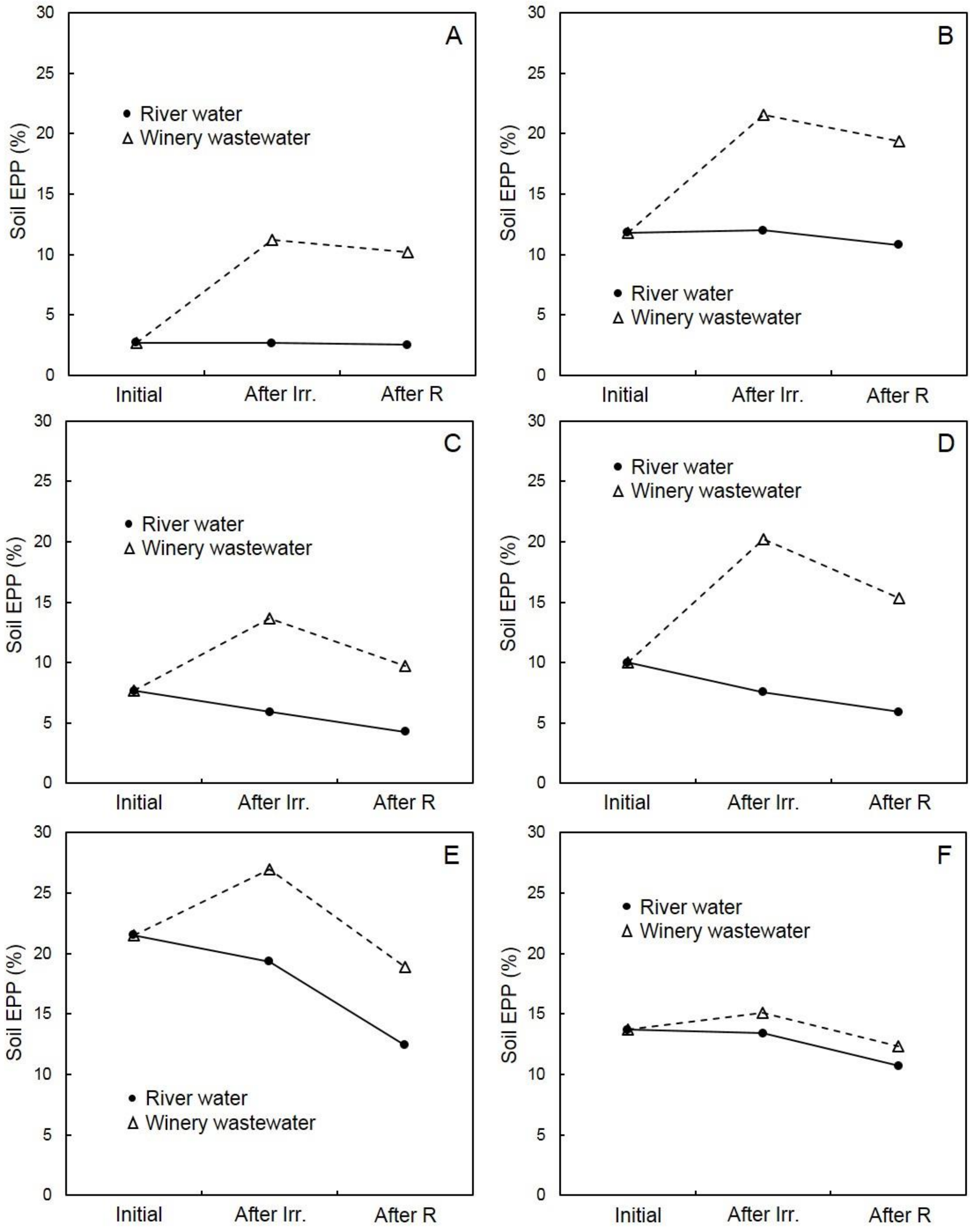

Figure 6.10. Effect of irrigation with clean water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on soil extractable potassium percentage (EPP') for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

<u>ESP':</u> In most soils irrigation with river water tended to increase the ESP' (Fig. 6.11). This was probably due to the increase in soil Na^+_{extr} (Fig. 6.7) upon irrigation with water containing a relatively high level of Na^+ (Fig. 6.2). Irrigation with diluted winery wastewater also increase the ESP'. At this stage there is no explanation why the ESP' showed the same increase in the Lutzville sand and the Robertson clay (Figs. 6.11B and 6.11F).

In the soils that were irrigated with river water, the simulated rainfall caused a decline in ESP' in most soils (Fig. 6.11). In the soils where leaching occurred, the ESP' was lower than the initial levels. Although Na^+ was the highest in the river water used for the simulation, the amount applied *via* the rainfall was considerably lower than the summed amount of the other cations (Tables 6.9 to 6.12). This probably also explains why the ESP' also tended to decrease in the soils where no leaching occurred, *i.e.* the Lutzville sand and the Robertson clay (Figs. 6.11B and 6.11F). In the soils that were irrigated with the diluted winery wastewater, the simulated rainfall caused a decline in ESP' in most soils (Fig. 6.11). In the soils where leaching occurred, the ESP' was also comparable to, or lower, than the initial levels before the irrigations were applied.

6.3.6.3. Soil $pH_{(KCl)}$

Irrigation with river water had no effect on the $pH_{(KCl)}$, irrespective of soil type (Fig. 6.12). This is in agreement with the results reported in Chapter 4. Irrigation with diluted winery wastewater tended to increase the $pH_{(KCl)}$ slightly, except in the Robertson clay (Fig. 6.12F). If soils are irrigated with diluted winery wastewater the $pH_{(KCl)}$ can increase substantially over time, as was shown in Chapter 4. The high amount of basic cations, particularly K^+ and Na^+, applied *via* the wastewater seems to be the reason for the $pH_{(KCl)}$ increase. In soils that were irrigated with river water, and where leaching occurred, the $pH_{(KCl)}$ tended to decline slightly (Fig. 6.12). In the Lutzville sand and Robertson clay where no leaching occurred, the $pH_{(KCl)}$ showed a slight incline upon the simulated rainfall (Figs. 6.12B & 6.12F). Since the K^+_{extr} showed almost no increase in the Lutzville sand and Robertson clay after the rainfall (Figs. 6.6B & 6.6F), the higher $pH_{(KCl)}$ was probably caused by the increase in soil Na^+_{extr} (Figs. 6.7B & 6.7F). The higher Ca^{2+}_{extr} (Fig. 6.8) and Mg^{2+}_{extr} (Fig. 6.9) after the simulated rainfall could also have contributed to the $pH_{(KCl)}$ increase in the Robertson clay.

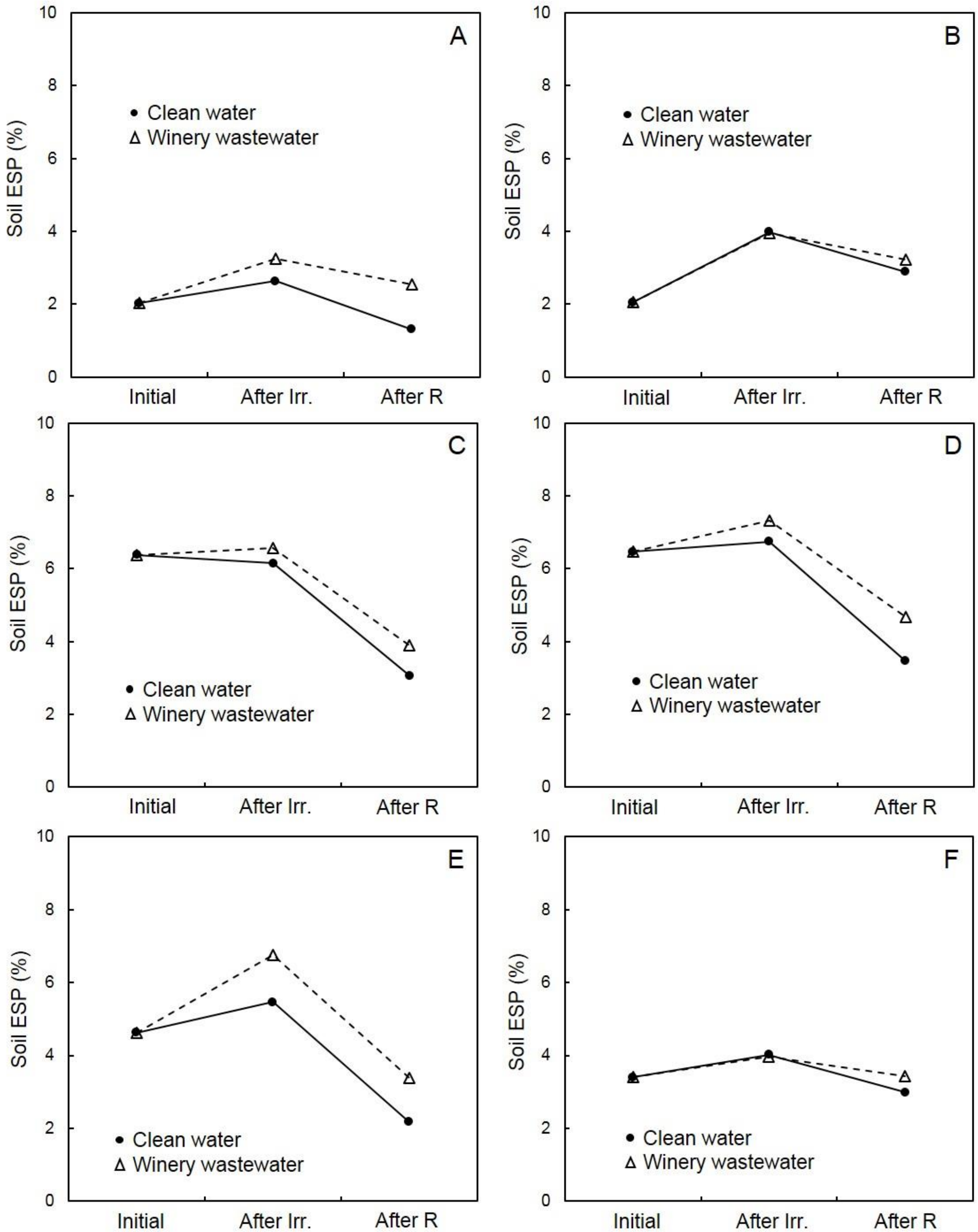

Figure 6.11. Effect of irrigation with clean water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on soil extractable sodium percentage (ESP') for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

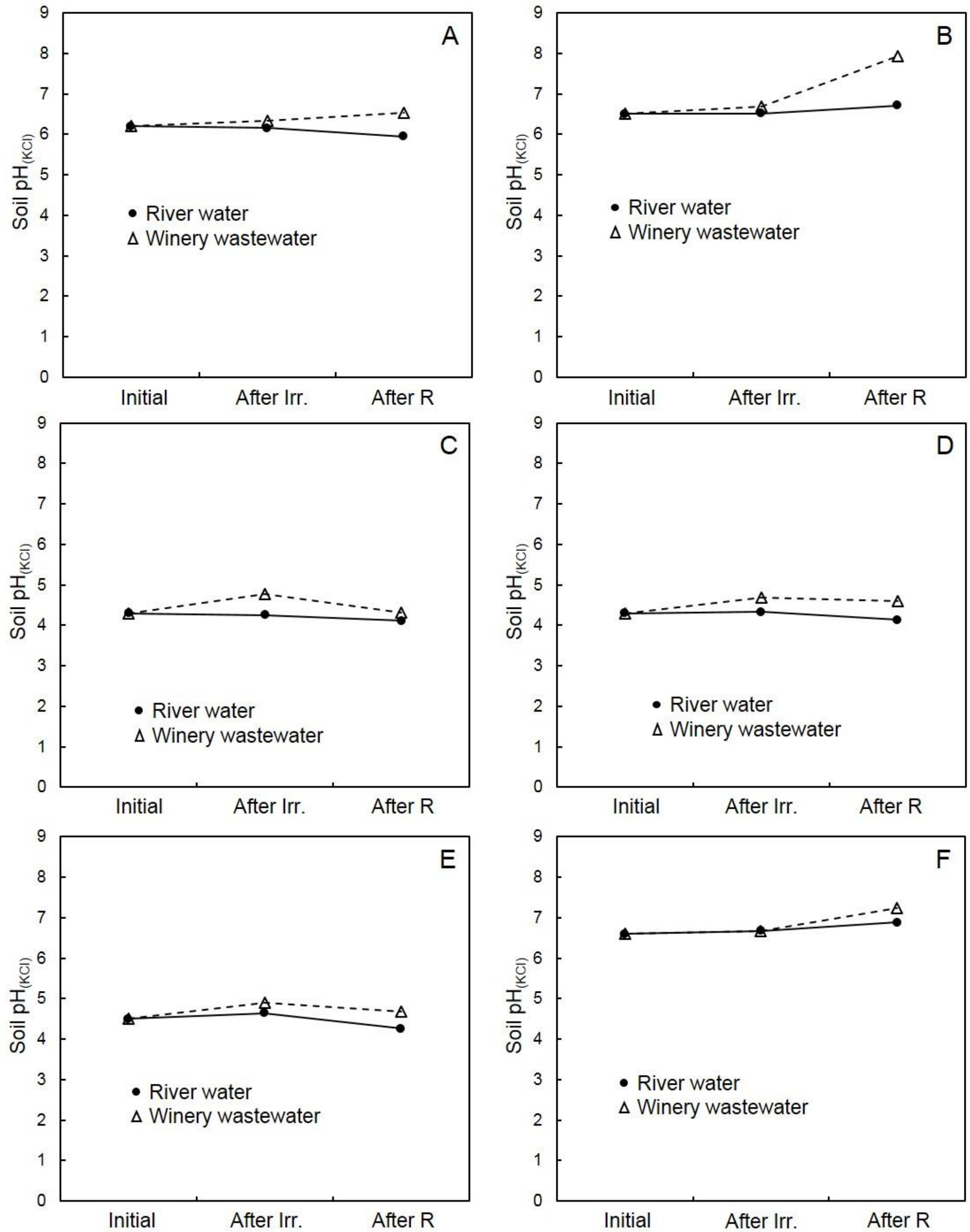

Figure 6.12. Effect of irrigation with clean water and winery wastewater diluted to 3000 mg/L COD (After Irr.) followed by simulated winter rainfall (After R) on soil $pH_{(KCl)}$ for (A) Rawsonville sand, (B) Lutzville sand, (C) Stellenbosch shale, (D) Stellenbosch granite, (E) Stellenbosch sand and (F) Robertson clay.

The diluted winery wastewater used for irrigation contained a high organic load compared to the river water (Tables 6.6 & 6.7). The cations in the wastewater were probably present in the form of organic salts. These salts can produce OH^- anions *via* decarboxylation that will increase the soil pH as illustrated in Figure 6.13 (Rukshana *et al.*, 2011). Organic acids present in the wastewater may also be a source of organic anions *via* the dissociation of H^+ which can increase the soil pH *via* decarboxylation (Fig. 6.13). If this happens, the soil might initially contain more H^+, but the pH will increase over time as more OH^- is formed (Rukshana *et al.*, 2011). The organic load in the wastewater could be a further source of organic N. These compounds will also produce OH^- anions which can increase the soil pH if ammonification occurs in soil (Fig. 6.13).

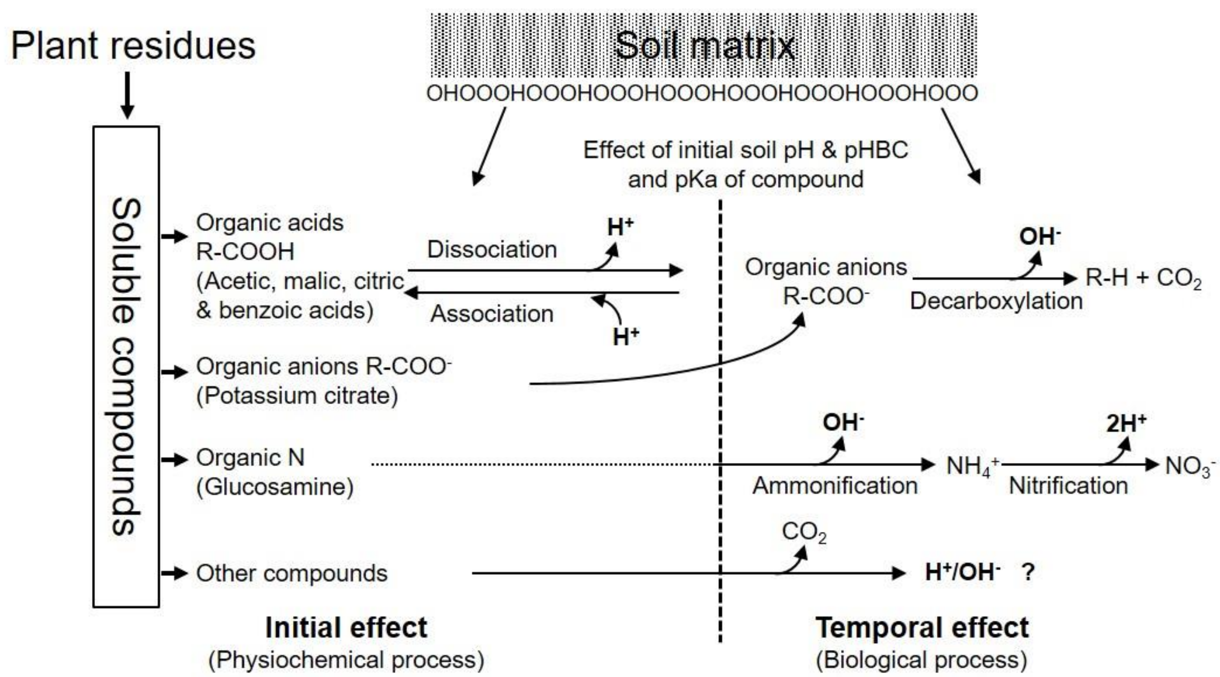

Figure 6.13. Diagram illustrating possible mechanisms of soil pH changes upon addition of model compounds (redrawn from Rukshana *et al.*, 2011).

6.4. CONCLUSIONS AND RECOMMENDATIONS

Irrigation with winery wastewater diluted to 3000 mg/L COD increased the basic cations which resulted in a slight $pH_{(KCl)}$ increase in the six soils, *i.e.* irrespective of clay content. The $pH_{(KCl)}$ increase was probably related to the soluble organic compounds in the wastewater. Leaching of cations, particularly, K^+ and Na^+, occurred only from four of the six soils when the winter rainfall was simulated. The total average K added to the soil was 2500 kg/ha while Na was 450 kg/ha. In one of the sandy soils, the simulated rainfall was too low to allow leaching. In the case of the other soil, a high clay content, *i.e.* 35%, in combination with low rainfall, prevented leaching. Where three soils received the same amount of rainfall, more cations leached from the sandy soil compared to the two heavier soils. These trends indicated that the leaching would be a function of soil texture, as well as rainfall. In fact, multiple linear regression models showed that the amount of exchangeable cations retained in the soils was a function of initial cation content, rainfall and organic carbon content. It must be noted that organic carbon content seemed to be a better indicator of the exchangeable cation amounts retained than clay content. This is most likely due the prominent effect of organic material on the CEC.

Since the leachate volumes were generally small, most of the cations applied were retained in the soils. The small volumes of leachate were due to the relatively low mean rainfall per simulated event. Therefore, it would have been more realistic to simulate the rainfall on a monthly, and not on a seasonal basis. However, the simulation with low rainfall events indicated that the exchangeable basic cations are more likely to accumulate in soils if climate change result in lower winter rainfall. Given highly variable rainfall in South Africa, it is also possible that abnormally high daily rainfall events can leach the accumulated cations from vineyard soils in regions with relatively low rainfall.

Due to the relatively low level of K^+ in the river water used for the rainfall simulation, the difference between the actual and estimated amount of K^+ retained, was less than 0.5%. In contrast, the amounts of Na^+ and Ca^{2+} retained, differed by as much as 13%, whereas the Mg^{2+} retained, differed up to *c.* 8%. Therefore, the Na^+, Ca^{2+} and Mg^{2+} in the water used for the simulations increased the actual amount of cations retained. Based on these results, it would be advisable to use genuine pure rainwater for leaching studies. It will be recommended that wineries should irrigate with wastewater based on the nutrient demand that the volume of the wastewater. Furthermore, care should be taken to avoid contamination of the harvested water, particularly if it needs to be stored in tanks.

CHAPTER 7. GENERAL CONCLUSIONS AND RECOMMENDATIONS

Specific conclusions have been included in each chapter. This chapter is aimed at providing general conclusions and recommendations to the industry and for future studies.

7.1. Scope of the study

The overall general aim of the study was to determine the suitability of selected soils from the Western Cape for winery wastewater irrigation in order to provide the South African wine industry with more knowledge regarding the response of different soil types found in the region to winery wastewater irrigation. The study investigated the effects of winery wastewater irrigation on (i) seasonal dynamics of soil chemical status, (ii) reactions of different soils and (iii) the role of winter rainfall in order to establish the risks of this practice on South African soils. Therefore, it distinguishes itself from any other studies carried out thus far, locally or internationally.

7.2. General conclusions

7.2.1. Seasonal soil chemistry dynamics due to winery wastewater irrigation on existing and new grazing paddocks

The study demonstrated that disposal of winery wastewater through land application, which is a general practice by South African wine farmers, causes high volumes of undiluted winery wastewater to be disposed of on very small areas. This practice results in over irrigation which aggravates leaching of large amounts of cations, particular K^+ and Na^+, beyond 90 cm soil depth. Unfortunately, the leached elements are bound to end up in natural water resources in the long run. To reduce this risk, wineries should be advised to apply the wastewater according to crop water requirements. As wastewater contains high amount of K, wastewater should also be applied based on the crop K requirement. The grazing paddocks used for disposal *via* irrigation should be as big as possible. Wineries should be encouraged to measure the quantity and quality of wastewater they dispose. The sprinklers used to irrigate the soil should be moved around regularly to avoid over irrigating beyond the soil's water holding capacity. The land application study confirmed that injudicious irrigation with undiluted winery wastewater poses a serious environmental hazard, particularly when the wastewater is applied to crops in sandy soils. Disposal of winery wastewater through land application can only be recommended where wastewater application will not exceed the water requirement of the grazing or other crop. Wineries should be advised to determine soil chemical status on a regular basis to minimise the risk of soil degradation. Proper winery wastewater management strategies such as irrigation scheduling should be considered for sustainable land application.

7.2.2. Effects of winery wastewater irrigation on the chemical properties of four different soils

A possible solution to reduce the abovementioned risk would be to dilute the winery wastewater before it is used for irrigation of vineyards or other crops. In this regard, a single mix and irrigation infrastructure made it possible to irrigate four different soils accurately with winery wastewater diluted to a COD of 3000 mg/L in a pot experiment. Irrigation with diluted winery wastewater increased the levels of extractable K^+ and Na^+ in the soils. The study confirmed that in clayey soils the increase of Na^+_{extr} with increasing amounts of Na^+ applied was almost double compared to the sandy soils. This indicated that the risk of Na^+ reaching excessive levels will be less where vineyards in sandy soils are irrigated with diluted winery wastewater than in clayey soils. Due to low Ca^{2+} and Mg^{2+} concentrations in the diluted winery wastewater, their concentrations in the soil at the end of the study were comparable to the initial levels. The application of winery wastewater led to soil $pH_{(KCl)}$ increases in all four soils irrespective of clay content. This could probably be attributed to organic anions added to the soil *via* irrigation with diluted winery wastewater.

7.2.3. Vulnerability of selected soils in the different rainfall areas to degradation and excessive leaching after wastewater application

Recommendations were developed regarding the suitability of winery wastewater irrigation in high and low rainfall areas respectively. Six different soils from three wine growing regions were subjected to simulated winter rainfall following one season of irrigation with winery wastewater. The winter rainfall could not leach basic cations, particularly K^+ and Na^+, from two of the six soils as the amount of the simulated rainfall was too low to achieve leaching. Where three soils received the same amount of rainfall, more cations leached from the sandy soil compared to the two clayey soils. These trends indicated that the leaching would be a function of soil texture, as could be expected, as well as rainfall. The simulation with low rainfall events indicated that the basic cations are more likely to accumulate in soils if climate change results in lower winter rainfall.

Given the highly variable rainfall in South Africa, it is possible that abnormally high daily rainfall events can leach accumulated cations from vineyard soils in regions with relatively low average rainfall, especially in sandy soils. In regions where winter rainfall is not high enough to leach nutrients from the soil that was irrigated with winery wastewater, there is a potential for accumulation of high salt concentrations that will increase soil salinity. It can be recommended that if the Lutzville sand and Robertson clay soils are to be used for wastewater irrigation, proper management and monitoring of soils are essential to avoid accumulation of salts due to low winter rainfall in these regions. The Stellenbosch duplex soil seems unsuitable for winery wastewater irrigation unless accurate irrigation scheduling is practised.

7.3. Recommendations

- Disposal of winery wastewater through land application can only be recommended where wastewater application does not exceed the water requirement of the crop.
- Wastewater needs to be distributed on an area of land that is big enough so that the daily applications do not cause over-irrigation.
- The effects of K: Na ratio in diluted or undiluted winery wastewater on soil structure stability, potassium availability and leaching of elements need to be addressed by continued research.
- Since climate, particularly rainfall, will affect the accumulation and/or leaching of elements, research should be carried out in field studies.
- Modelling studies to predict soil suitability and optimum level of winery wastewater dilution are essential to avoid pollution as a result of excessive leaching.
- Due to known lesser negative effect of K^+ on soil structure when compared to Na^+, it is recommended that will be advisable for winemakers to switch from sodium hydroxide detergents to potassium or ammonium hydroxide.
- The reason(s) why irrigation with diluted winery wastewater did not increase the soil organic carbon content must be investigated.
- Determination of the chemical status of permanent crops, *e.g.* vineyards, irrigated with diluted winery wastewater should be carried out at least annually.
- The soil chemical status should be determined at least annually. Soil samples must be collected as deep as practically possible to make sure that elements applied *via* the winery wastewater do not accumulate below the root zone and do not leach into streams and other water bodies.

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APPENDIX 3.1.

Description for a modal profile at an existing grazing paddock in Rawsonville.

Water table > 90 cm	**Slope**	**Erosion**	**Flood occurrence: Occasional**
Terrain unit: Valley bottom	Percentage: 1	Wind erosion: Slight	**Microrelief: none**
	Type: straight	Water erosion: slight	**Surface covering: None**
		Erosion stability: not stabilized	
Parent Material	**Weathering of underlying material**	**Alteration of underlying material: generalized**	
Lithology of solum:	Physical: Weak	**Vegetation/Land use: Vineyards**	
Origin: Single	Chemical: Weak		
Mode Alluvium			
Lithology of underlying material: Unknown			
SIOL HORIZON RECORD			
HORIZON	**Lower depth**	**Colour: dry**	**Colour: moist**
A	10cm	10YR 5/1	10YR 3/2
E	40cm	10YR 5/1	10YR 3/2
B	40cm+	10YR 7/1	10YR 3/1
Field estimated texture: less than 5% clay	**MOTTLES**	**SOIL STRUCTURE**	**CONSISTENCE**

	Mottles A: few <2%	Primary: Apedal	Dry: Loose
	Size: Fine <5mm	Size: Fine	Moist: Loose
	Contrast: Distinct		Wet stickiness: Non-sticky
	Colour: Brown		Wet plasticity: Non-plastic
MACROPORES & CRACKS	**Cementation of Horizon: none**	**Surface & Subsurface features**	Roots: Many
Very fine & Fine pores: Few	**Freelime- non hardend: none**	Kind: Bleached surface crust	Transition: Diffuse
Surface coating: Normal	Slickensides: none	**Depositional Stratification: none**	Topography: Smooth
Cracks: None	Cutans: none		
	Coarse fragments: none		
DIAGNOSTIC HORIZONS AND MATERIAL	SOIL FORM: **LONGLANDS**		
Orthic A Horizon	**FAMILY: 1000 SHERBROOK**		
E Horizon			
Soft Plinthic B horizon			

APPENDIX 3.2.

Description for a modal profile from the disposal area at a new grazing paddock in Stellenbosch.

Water table < 90 cm	**Slope**	**Erosion**	**Flood occurrence: Occasional**
Terrain unit: Lower footslope	Percentage: 1	Wind erosion: none	**Microrelief: none**
	Type: straight	Water erosion: none	**Surface covering: None**
		Erosion stability: stabilized	
Parent Material	**Weathering of underlying material**	**Alteration of underlying material: Kaolinised**	
Lithology of solum	Physical: Weak	**Vegetation/Land use: Cultivated pastures**	
Origin: Binary suspected	Chemical: Strong		
Mode Alluvium			
Lithology of underlying material: mixed lithology			
Soil Horizon Record			
Horizon	**Lower depth**	**Colour: dry**	**Colour: moist**
OB (Overburdened)	10cm	10YR 6/2	10YR 7/1
A	20cm	10YR 6/2	10YR 7/1
E	20-70cm	10YR 6/2	10YR 7/1
G	70+	10YR 6/2	10YR 7/1

Field estimated texture: less than 5% clay	**MOTTLES**	**SOIL STRUCTURE**	**CONSISTENCE**
	Mottles A: few <2%	Primary: Apedal	Dry: Hard
	Size: Fine <5mm	Size: Fine	Moist: Friable
	Contrast: Faint	Type: Single grain	Wet stickiness: Non-sticky
	Colour: Red and Brown		Wet plasticity: Non-plastic
MACROPORES & CRACKS	**Cementation of Horizon: none**	**Surface & Subsurface features**	Roots: Many
Very fine & Fine pores: Few	**Freelime- non hardend: none**	Kind: None	Transition: Abrupt
Surface coating: Normal	Slickensides: none	**Depositional Stratification: none**	Topography: Smooth
Cracks: None	Cutans: none		
	Coarse fragments: none		
DIAGNOSTIC HORIZONS AND MATERIAL	**SOIL FORM**: **Kroonstad**		
Orthic A Horizon	**FAMILY: 1000 Morgendal**		
E Horizon			
G			

APPENDIX 6.1

Average rainfall data for weather stations situated in the areas where the soils included in the study where collected.

Lutzville weather station		1971 - 1989 (18 years)		
Month	**Rainfall (mm)**	**No. of rainfall days per month**	**Rainfall per day (mm)**	**Intervals**
May	17.9	4	4.5	7.8
June	24.8	5.4	4.6	5.6
July	19.6	5.4	3.6	5.5
August	19.8	6	3.3	5.2
September	11.5	4.1	2.8	7.3
Total	93.6	24.9		

Rawsonville weather station		2000 - 2012 (12 years)		
Month	**Rainfall (mm)**	**No. of rainfall days per month**	**Rainfall per day (mm)**	**Intervals**
May	116.5	8.8	13.2	3.5
June	127.5	7.9	16.1	3.8
July	100.5	8.5	11.8	3.6
August	121.2	8.2	14.8	3.8
September	82.7	7.1	11.6	4.2
Total	548.4	40.5		

Stellenbosch weather station		1967 - 1989 (22 years)		
Month	**Rainfall (mm)**	**No. of rainfall days per month**	**Rainfall per day (mm)**	**Intervals**
May	106.8	10.4	10.3	3.0
June	108.7	10.5	10.4	2.9
July	110.4	10.6	10.4	2.9
August	86.5	10.1	8.6	3.1
September	56.7	8.8	6.4	3.4
Total	469.1	50.4		

Robertson weather station		1954 – 1989 (35 years)		
Month	**Rainfall (mm)**	**No. of rainfall days per month**	**Rainfall per day (mm)**	**Intervals**
May	32.5	6.8	4.8	4.6
June	31.9	6.9	4.6	4.3
July	26.9	6.5	4.1	4.8
August	41.8	7.8	5.4	4.0
September	19.8	5.8	3.4	5.2
Total	152.9	33.8		