Long-term Effects of Irrigation with Treated Municipal Wastewater on Soil Chemical and Physical Responses in Commercial Vineyards in the Coastal Region of South Africa

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The prolonged drought in the Western Cape province of South Africa in recent years has been particularly detrimental to the wine industry. Water restrictions imposed by the authorities, and the limited supply of fresh water that can be stored, have emphasised the need for alternative water sources for vineyard irrigation. Treated municipal wastewater has been used successfully as an alternative source of irrigation water in other countries. A long-term trial was conducted in commercial vineyards in the Coastal region of South Africa to assess the impact of treated municipal wastewater irrigation on vineyards. Cabernet Sauvignon and Sauvignon blanc grapevines were irrigated using treated municipal wastewater from the Potsdam wastewater treatment works for 11 years. Grapevines were either rainfed (RF), irrigated with treated municipal wastewater *via* a single dripper line (SLD), or received twice the volume of wastewater *via* a double dripper line (DLD). Irrigation using treated municipal wastewater increased soil $pH_{(KCl)}$, the electrical conductivity of the saturated extract (EC_e) and Cl^-. Substantial amounts of Na^+ and K^+ accumulated in the topsoil due to irrigation with treated municipal wastewater. These soil K^+ increases could have a negative effect on wine colour stability should the levels of soil K^+ be such that they are absorbed excessively by grapevines. The near-saturation hydraulic conductivity (K_{ns}) at the surface of the soil could be related to the EC_e in the topsoil. The results represent specific in-field situations in three commercial vineyards under one set of climatic conditions.

INTRODUCTION

There have been frequent water shortages and below-average rainfall in the Western Cape of South Africa recently that have led to the worst drought in the province in recent memory. The drought was especially detrimental to the wine industry due to the carry-over effects of water constraints on grapevine growth and yield. Therefore, water scarcity is an increasingly important challenge for the viticultural sector in the region. These challenges have emphasised the need for alternative irrigation water sources. One such alternative could be treated municipal wastewater, which has been used as a source of irrigation water in many arid and semi-arid countries (Levy *et al*., 2014). In South Africa, approximately 2 000 ha of vineyards in the Swartland region are being irrigated with treated municipal wastewater from the City of Cape Town's Potsdam wastewater treatment works (WWTW) and the Malmesbury municipality (Myburgh, 2018). However, aside from the current study, there have been no other studies that have assessed the feasibility of using treated municipal wastewater for vineyard irrigation under South African conditions.

The irrigation of agricultural crops with municipal wastewater was recently reviewed by Hoogendijk *et al*. (2023a). In summary, there are possible benefits of using treated municipal wastewater for vineyard irrigation, as it can provide an extra source of water for irrigation that can improve grapevine growth and yield. Since domestic water sources often contain high amounts of macro-elements (Hoogendijk, 2019), nutrients such as nitrogen (N), phosphorous (P) and potassium (K^+) can be recycled if applied *via* the irrigation water. Organic compounds present in the treated municipal wastewater may have

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positive effects on soil structural stability. However, the high levels of salts in treated municipal wastewater can affect the physical, chemical and biological properties of the soil. Sodium (Na^+) and K^+ are particularly detrimental in terms of soil structural stability and increasing soil salinity. The presence of large amounts of monovalent cations could also result in clay dispersion, which can subsequently clog soil pores and limit water movement into and through the soil. In addition, irrigation using K^+-rich wastewaters may lead to excessive K^+ uptake by grapevines, which potentially can have a negative effect on wine quality (Laurenson *et al*., 2012).

Although there is extensive literature available on the effect of irrigation with municipal wastewater on soil chemical properties (Hermon, 2011 and references therein; Hoogendijk *et al*. 2023a), there is very little information regarding the re-use of municipal wastewater for vineyard irrigation. In this regard, soil samples from vineyards in McLaren Vale, South Australia that were irrigated by means of drip irrigation with either mains water or municipal wastewater for four to 11 years indicated that irrigation with municipal wastewater increased soil Na^+ and Mg^{2+}, but reduced soil Ca of a deep sand, clay loam and a hard-setting sandy loam soil (Laurenson, 2010). Increased soil salinity as a response to irrigation with municipal wastewater was reported for a vineyard in South Australia (McCarthy, 1981). In Spanish vineyards irrigated by means of drip irrigation, the use of treated municipal wastewater increased soil electrical conductivity of the saturated extract (EC_e), N, K^+, Na^+, magnesium (Mg^{2+}) and manganese (Mn^{2+}) compared to well water (De las Heras & Mañas, 2020). Treated municipal wastewater was used for vineyard irrigation in Australia and compared to freshwater irrigation where it was applied (Hermon, 2011). Pastures or areas adjacent to the vineyards and which were not irrigated were also sampled in a fence-line experimental design. The topsoil pH and EC were higher where treated municipal wastewater was used for irrigation, in comparison to irrigation with fresh water and in unirrigated pastures adjacent to the vineyards. The soil exchangeable sodium percentage (ESP) also increased, particularly in the soil layers up to 60 cm below the soil surface.

Taking the above-mentioned into consideration, the objective of this study was to assess the effects of long-term irrigation with treated municipal wastewater on soil chemical and physical properties in commercial vineyards in the Coastal region of the Western Cape.

MATERIALS AND METHODS

Site selection and vineyard characteristics

The field trial was carried out in commercial vineyards on a farm near the town of Philadelphia in the Western Cape, where three experimental sites were selected in different landscape positions. Details on site selection, vineyard characteristics and grapevine responses were reported previously (Howell *et al*., 2022; Hoogendijk *et al*., 2023b).

Irrigation treatments and application

Details on the irrigation treatments and application were reported previously (Howell *et al*., 2022; Hoogendijk *et al*., 2023b). In summary, there were three different irrigation treatments. The first treatment was rainfed (RF), which was considered to be a control treatment as no raw water was available for vineyard irrigation on the farm. The second treatment was drip irrigated with treated municipal wastewater *via* a single dripper line (SLD) in the grapevine row. The third treatment had a double dripper line (DLD), which supplied double the volume of wastewater in the grapevine row.

Irrigation water origin and quality

An assessment of the water quality and nutrient load of the treated municipal wastewater applied in the study was reported by Howell *et al*. (2022).

Soil chemical properties

Baseline soil samples were taken in 2006 before wastewater irrigation commenced (ARC, unpublished data). Following 11 years of irrigation with treated municipal wastewater, samples were taken at budbreak (September) of the 2017/2018 season. Soils were sampled in 30 cm increments to a depth of 90 cm in all plots, and up to 180 cm in all treatment plots where it was possible to sample deeper. Soil chemical analyses were carried out by a commercial laboratory. The $pH_{(KCl)}$ was determined in a suspension of 1 M potassium chloride (KCl) at 25°C. Soil electrical resistance of the saturated paste extract (R_s) was determined according to methods presented by Jones (1999). Thereafter, R_s was converted to EC_e using the following equation:

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

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

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

$$ESP' = (Na^+ \div S) \times 100 \quad \text{(Eq. 2)}$$

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

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$$EPP' = (K^+ \div S) \times 100 \quad \text{(Eq. 3)}$$

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

Bray II P and K^+ were determined by extraction with 0.03 M ammonium fluoride (NH_4F) in 0.01 M hydrochloric acid (HCl). The P and K^+ concentrations in the extract were determined in the same manner as the basic cations. Soil organic carbon (SOC) was determined using methods described by Walkley and Black (1934). Soil chloride (Cl^-) was not determined at the beginning of the larger study period. During the 2017/2018 season, soil Cl^- was determined volumetrically *via* titration of a 0.1 M potassium nitrate (KNO_3) soil extract with 0.043 M silver nitrate ($AgNO_3$), using potassium dichromate (K_2CrO_4) as an indicator (Chapman & Pratt, 1961).

Soil physical properties

Soil texture

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

Near-saturation hydraulic conductivity

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

$$K_{ns} = \{[(V_i - V_f) \div 1000] \div 0.001521\} \times 60 \div \Delta t \quad \text{(Eq. 4)}$$

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

Statistical analysis

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

RESULTS AND DISCUSSION

Rainfall

The monthly rainfall for the 2017/2018 season measured from July 2017 to June 2018 in relation to the long-term mean (LTM) measured over the 12-year study period is shown in Fig. 1. The rainfall during the 2017/2018 season was below the LTM for most of the season and only exceeded the LTM during August 2017, as well as in April and May 2018 (Fig. 1). An average of 86 mm was measured during the summer months of September to March throughout the study period, whilst only 39 mm was measured during the summer of 2017/2018. The average winter rainfall from May to August was 160 mm. The LTM annual rainfall measured from July to June of each year was 253 mm, while only 118 mm was measured from July 2017 to June 2018.

Soil texture

The soil texture at the shoulder site was relatively uniform and was predominantly clay to clay loam, with a clay content that ranged between 35% and 51%. The soils at this experimental site had high stone fractions that ranged between 15.3% and 45.8%. This was mainly due to the presence of the shale parent material, which was brought up to the soil surface when the soil was prepared before planting. The backslope

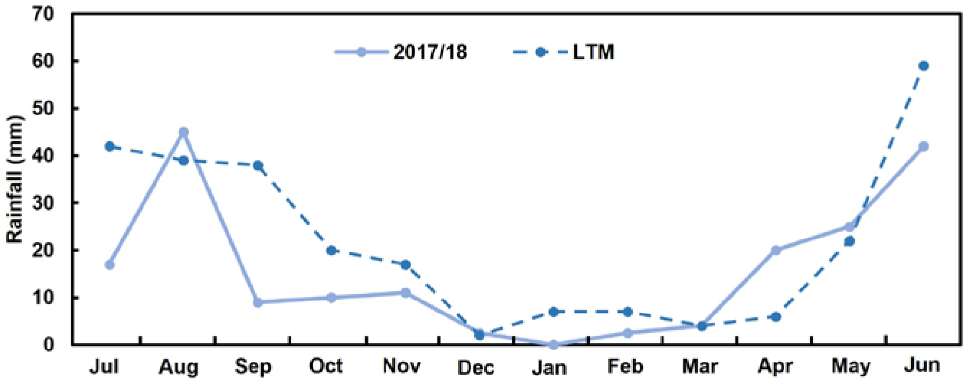

FIGURE 1

Monthly rainfall (mm) from July 2017 to June 2018 in relation to the long-term mean (LTM) measured over the 12-year study period on a farm near Philadelphia (ARC, unpublished data).

site had a sandy loam topsoil with a clay content of 15% to 19%, and a sandy clay loam to clay layer. The clay content ranged between 29% and 49% at 90 cm. The soil of the backslope did not contain any stones. The 0 cm to 30 cm layer of the footslope site ranged from sandy loam in the RF and SLD plots to sandy clay loam in the DLD plot. The clay content of the footslope DLD plot was between 17% and 25% in the 0 cm to 30 cm layer, and increased with depth to a maximum of 55% at 120 cm. The only presence of stones at the footslope site was in the 30 cm to 60 cm soil layer of the SLD plot. As expected, the soils differed substantially between the three landscape positions. However, soils were relatively uniform between treatment plots at the individual landscape positions.

Soil chemical properties

$pH_{(KCl)}$

The $pH_{(KCl)}$ of the 0 cm to 30 cm soil layer increased in all of the landscape positions following 11 years of irrigation with treated municipal wastewater (Fig. 2A to 2C). At the shoulder site, the DLD plot had the highest topsoil pH at the end of the study period, followed by the SLD and RF plots (Fig. 2A), although the differences between the three treatments were relatively small. In contrast, the topsoil pH of the SLD and DLD plots on the backslope site was similar, whilst the RF was considerably lower (Fig. 2B). The topsoil pH of the backslope SLD plot increased by approximately 2.3 units compared to the baseline before irrigation commenced. Similar results were obtained at the footslope site (Fig. 2C). At this site, the topsoil pH of the RF treatment remained unchanged, but decreased with depth when compared to the baseline.

On average, the topsoil pH of the SLD and DLD plots increased by 1.3 units after 11 years of irrigation with treated municipal wastewater (Fig. 3). The increase in pH was most likely due to the pH of the irrigation water, which varied between 6.7 and 8.0 throughout the study period. The decarboxylation and hydrolysis of organic acids and bicarbonate anions present in the wastewater may also have contributed to the increased pH (Li *et al*., 2008). The increased pH did not cause concern, as it remained near neutral and would therefore have had little effect on biological functioning (Schipper *et al*., 1996). In addition, the pH of all of the treatment plots was within the recommended range to sustain optimal grapevine growth, viz. 5.0 to 7.5 (Saayman, 1981). Similar results have been reported by Sparling *et al*. (2006) and Schipper *et al*. (1996) where soils were irrigated with secondary and tertiary treated municipal wastewater, respectively. In contrast, Xu *et al*. (2010) reported a decrease in soil pH of *ca.* 1.1 units in a 150 cm soil profile following 20 years of irrigation with treated municipal wastewater.

EC_e

The EC_e of the 0 cm to 30 cm soil layer increased in all of the treatment plots at each of the landscape positions compared to the baseline values before irrigation commenced (Fig. 4A to C). The topsoil EC_e of the shoulder DLD plot was nearly three times higher than the baseline value (Fig. 4A). In addition, it was the highest EC_e measured across all three landscape positions. However, no clear trends could be observed in the subsoil between the treatments of the shoulder site. On the backslope site, the SLD treatment had the highest topsoil EC_e. However, the differences between the treatments and the baseline measurement were relatively small (Fig. 4B). An increase in EC_e with soil depth was also evident at this site. The topsoil EC_e of the footslope DLD plot increased from 0.17 dS/m before irrigation with treated municipal wastewater began to 0.56 dS/m following 11 years of irrigation (Fig. 4C).

The mean topsoil EC_e increased with the amount of treated municipal wastewater applied (Fig. 5). However,

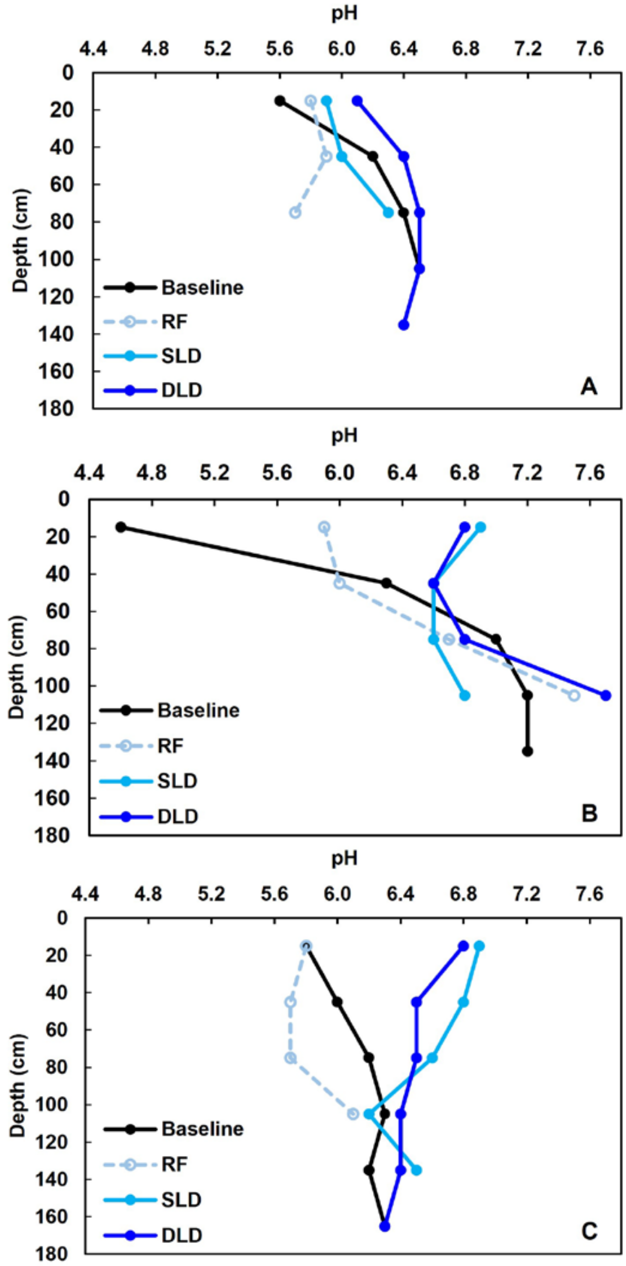

FIGURE 2
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil pH on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

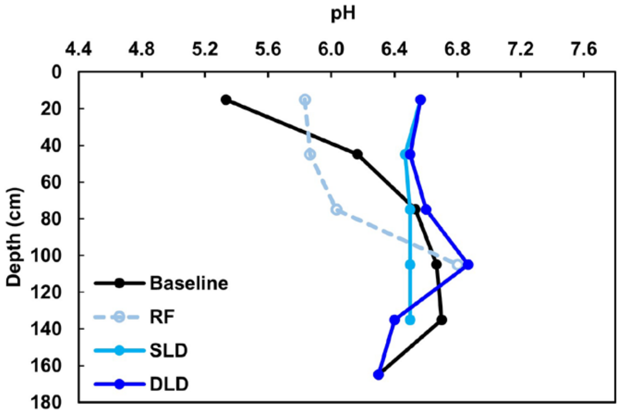

FIGURE 3
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean soil pH across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

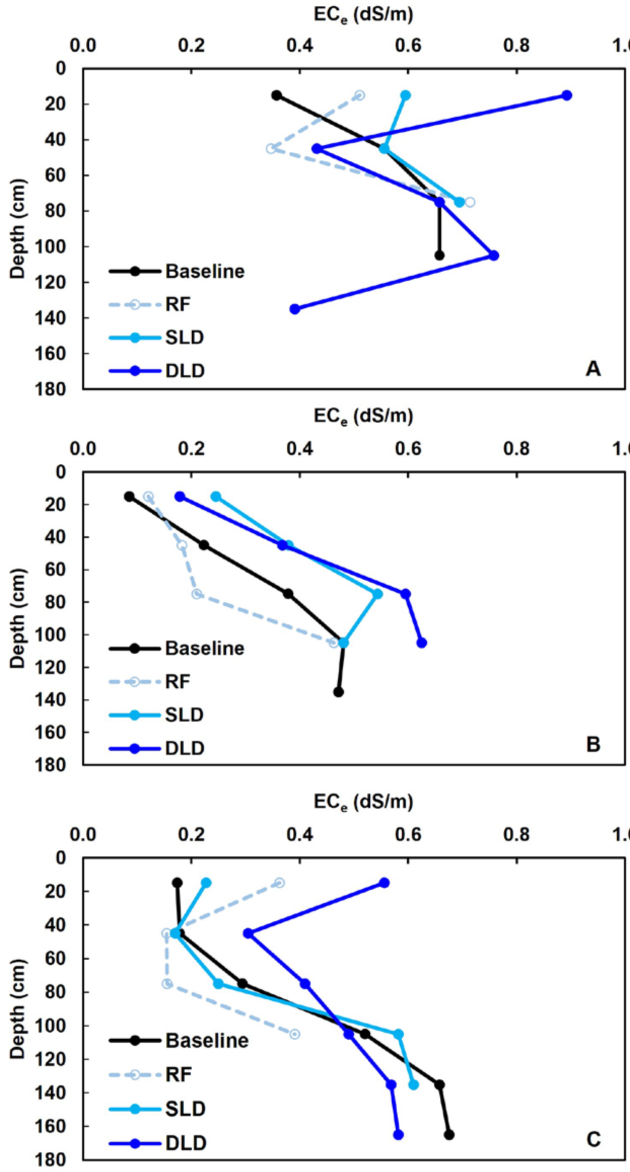

FIGURE 4
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil electrical conductivity (EC_e) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

there were no clear trends in the deeper soil layers that could be related to the different irrigation treatments compared to the baseline values. The increased EC_e of the topsoil indicates an accumulation of salts at the soil surface. The EC_w of the treated municipal wastewater ranged between 0.7 dS/m and 1.2 dS/m (Howell *et al*., 2022), which could explain the increased EC_e. The accumulation of salts at the soil surface is most likely a result of high evapotranspiration during the irrigation season, which concentrated the salts in the upper parts of the root zone (Rhoades *et al*., 1973). Similar results were reported for vineyard soils in Great Western, Australia that were irrigated with treated municipal wastewater for at least five years (Hermon, 2011). The accumulation of salts in the soil profile is of concern, as a progressive increase in soil salinity can result in deficiencies in grapevine nutrients (McCarthy, 1981; Paranychianakis *et al*., 2006). However, the relatively small increase in EC_e after 11 years of wastewater irrigation in the current study suggests that winter rainfall might have leached some of the applied salts beyond the measured depth. In a laboratory study in which rainfall cycles were simulated, the EC of the drainage water was considerably higher than that of the input water (Laurenson, 2010), indicating a loss of salts from the soil during rainfall events. Therefore, regular rainfall events may help to alleviate high soil EC_e where municipal wastewater containing high levels of salts is used for irrigation.

Bray II P

Bray II P increased in the 0 cm to 30 cm soil layer at all of the experimental sites following 11 years of irrigation with treated municipal wastewater (Fig. 6A to 6C). The soil Bray II P content at the shoulder site was highest at the SLD, followed by the RF plot, whilst the Bray II P content of the DLD plot was approximately half that of the SLD (Fig. 6A). The SLD and RF plots at the shoulder site exceeded the norm of 30 mg/kg P recommended for grapevines in soils containing more than 15% clay (Conradie, 1994). The Bray II P concentration in the 0 cm to 30 cm soil layer of the SLD and DLD plots at the backslope site was slightly less than the RF (Fig. 6B); however, all plots met the norm of 30 mg/kg. The RF plot at the footslope site had the highest Bray II P content in the 0 cm to 30 cm soil layer compared to the other treatments (Fig. 6C). The P content of the RF plot increased from 7 mg/kg measured before the study commenced to 46 mg/kg 11 years thereafter.

After 11 years of irrigation, the soil contained on average, 42.6 mg/kg, 39.3 mg/kg and 28.1 mg/kg Bray II P in the RF, SLD and DLD treatments, respectively, whilst the baseline value was 12.7 mg/kg. An accumulation of P in the topsoil was evident, as the concentration decreased sharply in the 30

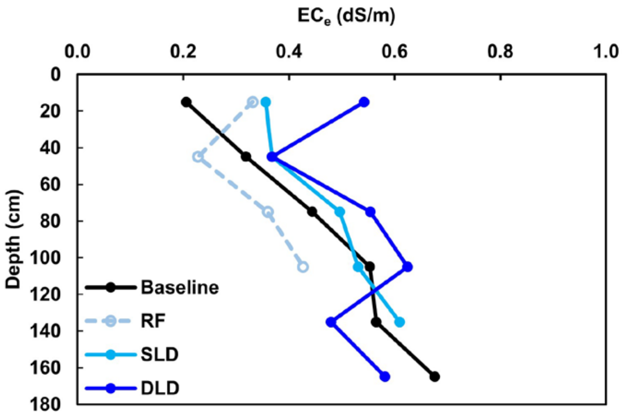

FIGURE 5

Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean soil electrical conductivity (EC$_e$) across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

cm to 60 cm soil layer in all of the treatments and remained at relatively constant levels in the 60 cm to 90 cm soil layer (Fig. 7). The 0 cm to 30 cm soil layers of the RF and SLD treatments consistently exceeded the norm of 30 mg/kg P for soils with a clay content of at least 15%. In contrast, the DLD treatments were on average only 2 mg/kg below the norm. The accumulation of P in the topsoil of the RF treatments could be explained by the application of P fertiliser by the grower and the low vigour that is expected of grapevines that are grown under dryland conditions, which would absorb very small amounts of P from the soil. In contrast, the high vigour that is expected from over-irrigated grapevines is reflected in the lower soil P content of the DLD plots, despite the application of additional P *via* treated municipal wastewater irrigation.

Bray II K

The Bray II K^+ content of the 0 cm to 30 cm soil layer of the shoulder site increased in all of the treatment plots compared to the baseline values (Fig. 8A). The slightly increased K^+ content of the RF plot on the shoulder site was due to the application of K^+ fertiliser by the grower, whereas the high K^+ content of the DLD plot may be the result of the low mobility of K^+ in the soil and its retention by clay minerals (Pérez *et al*., 2015), as well as an over-supply of K^+ *via* treatment with treated municipal wastewater. On the backslope site, the K^+ content of the 0 cm to 30 cm soil layer did not differ substantially between treatment plots, but increased in all of the plots compared to the baseline (Fig. 8B). The K^+ content of the SLD and DLD plots was maintained in the deeper soil layers, whereas a gradual decrease up to 90 cm was observed in the RF plot. Bray II K^+ levels in the 90 cm to 120 cm soil layer of the backslope site decreased in all of the treatment plots compared to the baseline. This was probably due to K^+ uptake by grapevines from deeper soil layers, or the leaching of K^+ beyond the measured depth. The K^+ content

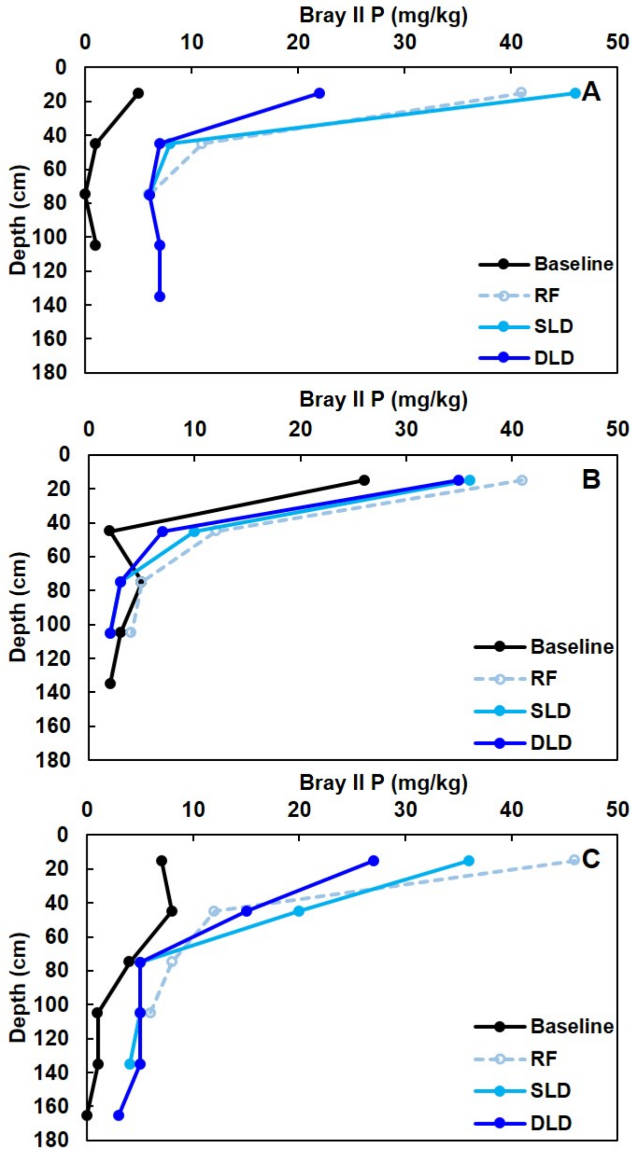

FIGURE 6

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

of the topsoil layer of the footslope increased under DLD compared to the baseline (Fig. 8C). Similar to the backslope site, K^+ levels beyond 90 cm decreased below the baseline levels in all of the treatments.

On average, the Bray II K^+ content of the 0 cm to 30 cm soil layer increased by 26 mg/kg, 42 mg/kg and 127 mg/kg for the RF, SLD and DLD treatments, respectively (Fig. 9). An accumulation of K^+ in the topsoil due to municipal wastewater irrigation has previously been reported by Heidarpour *et al*. (2007), Kiziloglu *et al*. (2007) and Singh *et al*. (2012). The high K^+ content under DLD is of concern, since an over-supply of K^+ to grapevines may result in excessive K^+ uptake. This could lead to musts with high pH and malate concentrations,

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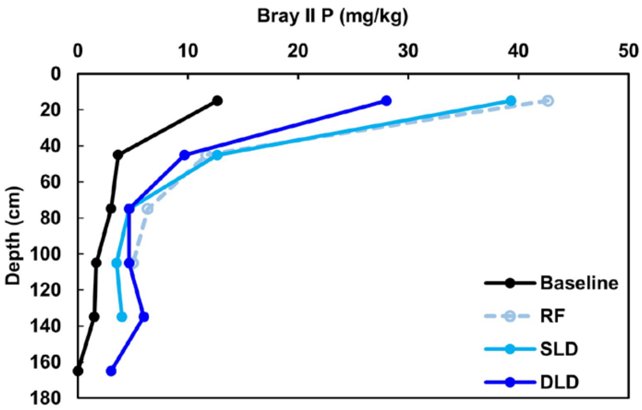

FIGURE 7
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean Bray II extractable phosphorous (P) content across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

as well as poor colour in red wines (Mpelasoka *et al*., 2003). An accumulation of K^+ in the soil can also have deleterious effects on soil structure (Laurenson *et al*., 2012), and affect soil *K* and IR negatively due to its effects on clay dispersal (Arienzo *et al*., 2009). However, clay dispersion is highly dependent on the electrolyte concentration of the infiltrating water (Shainberg *et al*., 1981). Therefore, the high salinity that is often associated with wastewater might mitigate the negative effects of K^+ on aggregate stability and *K* (Arienzo *et al*., 2009). However, as long as the EC_w of the treated municipal wastewater remains above the critical coagulation value, no soil dispersion is expected to occur (Abedi-Koupai *et al*., 2006).

Extractable Ca^{2+} and Mg^{2+}

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

Similar to Ca^{2+}, extractable soil Mg^{2+} levels did not show any consistent trends at the various landscape positions that could be related to the different irrigation treatments (data not shown). The mean soil Mg^{2+} concentrations of the 0 cm to 30 cm soil layer in the RF and SLD treatments decreased when compared to the baseline values, whilst the DLD

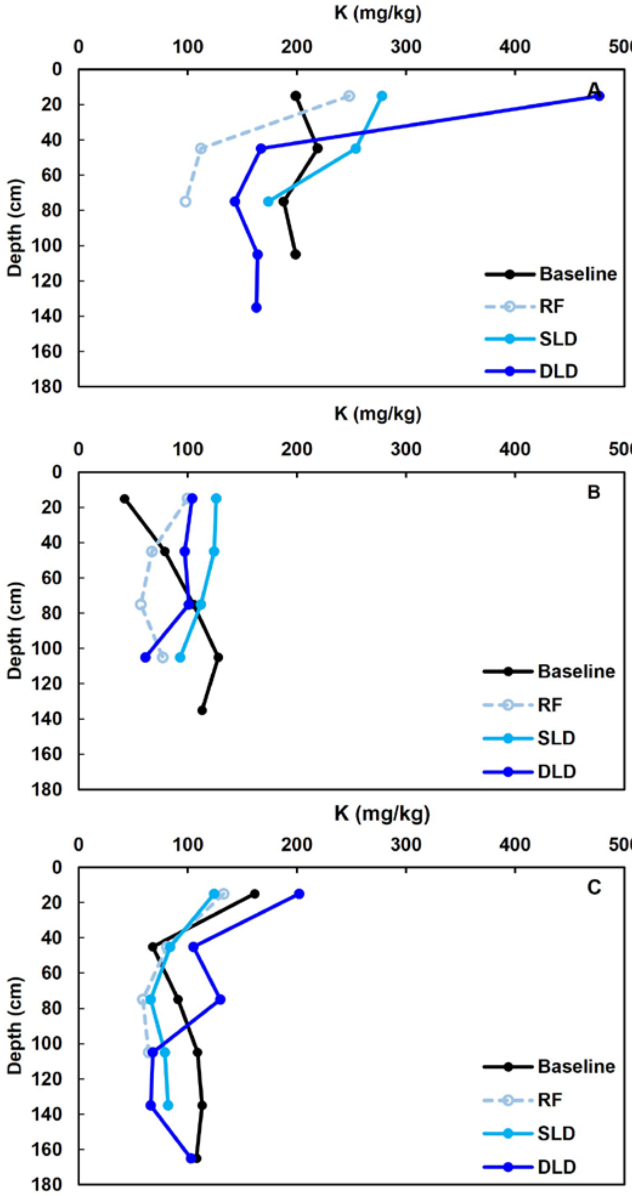

FIGURE 8
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil Bray II extractable potassium (K) content on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

remained relatively unchanged after 11 years of irrigation using treated municipal wastewater (data not shown). This is probably due to small amounts of Mg^{2+} supplied to the soil and the uptake of Mg^{2+} by grapevines, which depleted soil Mg^{2+} levels over the long term. Neilsen *et al*. (1991) attributed the reduction in soil Mg^{2+} following five years of municipal wastewater irrigation to mass exchange by Na^+ and K^+.

EPP′

Soil extractable K^+ followed similar trends as observed for Bray II K^+ (data not shown), therefore they will not be

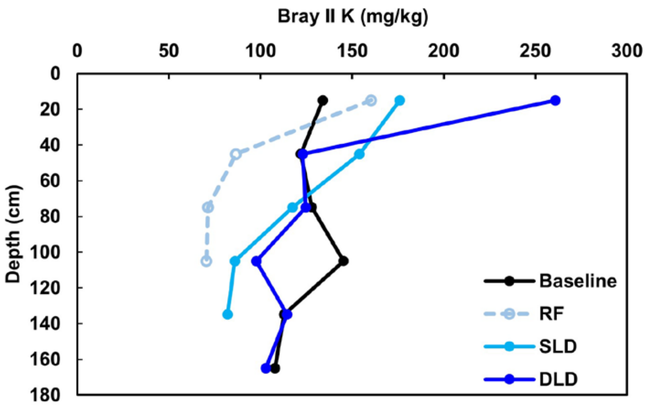

FIGURE 9
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean Bray II extractable potassium (K) content across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

discussed. Since exchangeable K^+ was not determined, the EPP′ was calculated rather than the EPP. Conradie (1994) recommended a ratio of 3% to 4% for exchangeable K^+ to other cations. The baseline EPP′ in the 0 cm to 30 cm soil layer of the shoulder site already exceeded this norm, and a further increase in all of the treatments was observed throughout the study period (Fig. 10A). These results were expected, as the topsoil of this particular site had a high clay content, which could retain a high amount of extractable K^+. Furthermore, the EPP′ of the DLD plot was ca. 2% higher than the SLD plot and 3% higher than the RF plot, indicating a steady increase in EPP′ due to irrigation with treated municipal wastewater. The EPP′ of the 0 cm to 30 cm soil layer of the backslope site increased in all the treatment plots compared to the baseline, with the RF plot having the highest EPP′, followed by the SLD and DLD plots (Fig. 10B). The accumulation of extractable K^+ on the RF plot could be explained by low vigour and grape production, which are associated with grapevines grown under dryland conditions, and the subsequent lower uptake of K^+ from the soil.

The topsoil EPP′ of the backslope DLD plot remained similar to the baseline, and the SLD plot only increased by 0.8% following 11 years of wastewater irrigation. The lack of response to wastewater irrigation in terms of EPP′ at this site might be explained by the uptake of K^+ by grapevines and the leaching of excess K^+ from the soil profile, as the clay content of this site is considerably lower than that of the shoulder site (data not shown). At the footslope site, the baseline EPP′ of the 0 cm to 30 cm soil layer was 10% and increased by 1% and 2% on the SLD and DLD plots, respectively, while the RF plot remained unchanged (Fig. 10C). The higher EPP′ was expected on the DLD plot of this site due to the larger volume of K-containing wastewater applied to this site. The EPP′ decreased with depth in all of the treatment plots, but increased in relation to the baseline in the 30 cm to 60 cm soil layer and reached levels below the baseline at a depth of 90 cm. The higher EPP′ in the 30 cm to 60 cm layer could be explained by the higher clay content of this layer. The decrease in deeper layers may be due to a combination of grapevine K^+ uptake and K^+ leaching from the soil profile.

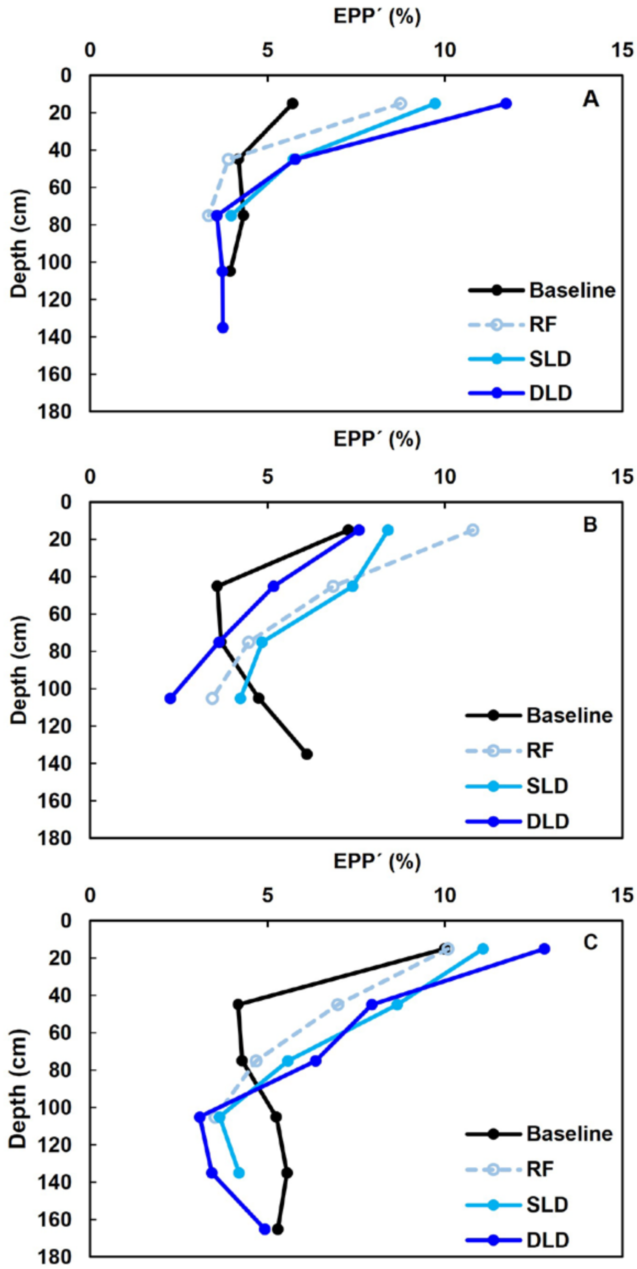

FIGURE 10
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil extractable potassium percentage (EPP′) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

The mean EPP′ for the baseline exceeded the recommended norm by far and increased substantially in all of the treatments throughout the study period (Fig. 11). There was little difference between the topsoil EPP′ of the RF and SLD treatments, but the DLD was 2% higher than in the other treatments. The high extractable K^+ at all of the experimental sites is concerning, as it could lead to greater K^+ uptake by the grapevines, which ultimately may result in unstable wines with a high pH (Gawel *et al*., 2000; Mpelasoka *et al*., 2003). In addition, high amounts of exchangeable K^+ have been associated with reduced *K* (Quirk & Schofield, 1955; Chen *et al*., 1983).

ESP´

Soil extractable Na^+ followed similar trends as those observed for ESP´ (data not shown), therefore only ESP´ is discussed. The ESP´ of the 0 cm to 30 cm soil layer of the shoulder site decreased in the RF plot, but levels similar to the baseline were maintained in the SLD and DLD plots (Fig. 12A). An increase in ESP´ was observed with depth in all of the treatment plots. However, the values were similar between treatment plots and lower than the baseline. An increase in ESP with depth in three Israeli soils irrigated with treated municipal wastewater with SAR similar to the present study was attributed to an increase in clay content at deeper soil layers (Levy *et al*., 2014). The reason for the high baseline ESP´ values in the subsoil layers of the shoulder site could be explained by the increased weathering of clay minerals due to soil preparation. Irrigation with treated municipal wastewater increased the ESP´ throughout the soil profile in the backslope site (Fig. 12B). However, little difference in Na^+ accumulation could be seen between the SLD and DLD plots. The values measured in the RF plot of the backslope site at the end of the study period remained comparable to the baseline. Similar results were observed when olive orchards were irrigated with treated municipal wastewater and compared to a rainfed control treatment (Ayoub *et al*., 2016).

The ESP´ of the footslope site followed a similar trend to that of the backslope site (Fig. 12C), with the RF plot remaining largely unaffected, and the SLD and DLD plots increasing substantially following 11 years of irrigation with treated municipal wastewater. However, the SLD plot on the footslope site had greater ESP´ values in the subsoil compared to the DLD plot, indicating more accumulation of salts in the subsoil under SLD. The increase in ESP´ in the backslope and footslope sites is particularly concerning, as the SAR of the irrigation water largely remained below 5, which is the critical limit for wastewater used for irrigation (Department of Water Affairs [DWA], 2013). Similarly, Levy *et al*. (2014) reported ESP levels in sandy clay subsoils (≥ 15% clay) reaching between 6% and 16% following ten years of irrigation with treated municipal wastewater with an SAR of 3 to 5.

The accumulation of Na^+ was attributed to a possible lack of chemical equilibrium between the SAR of the treated wastewater, the SAR of the soil solution and the ESP of the subsoil layers. The replacement of exchangeable Na^+ (applied *via* wastewater irrigation) in the topsoil by Ca^{2+} originating from the dissolution of calcium carbonate during the rainy season, and the subsequent leaching of Na^+ to deeper soil layers where it is re-adsorbed to the soil exchange complex, was proposed as a possible mechanism for the Na^+ accumulation (Levy *et al*., 2014). Results presented by Myburgh (2018) indicated that Na^+ accumulation in the

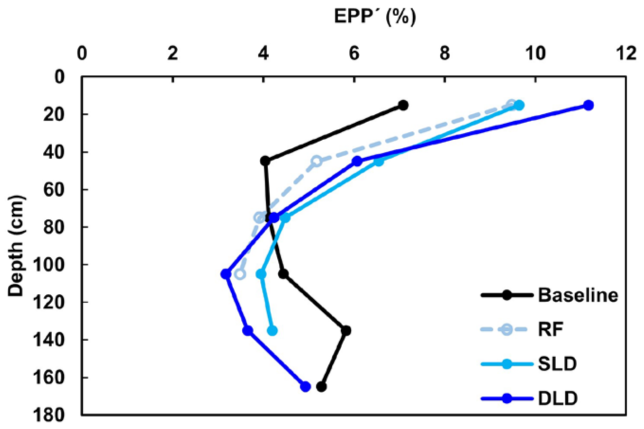

FIGURE 11
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean soil extractable potassium (EPP´) across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

soil profiles of these sites was highly dependent on winter rainfall, implying that salts will accumulate in the soil during winters with low rainfall and leach to deeper layers following higher rainfall. The winter rainfall preceding the collection of the soil samples in 2017 was 138 mm (data not shown), which was lower than the LTM of 160 mm, and therefore could have contributed to the accumulation of Na^+ in the wastewater-irrigated sites.

On average, the topsoil ESP´ increased by 1% and 3% for the SLD and DLD treatments, respectively, whereas the RF treatment was 3% lower compared to the baseline (Fig. 13). A steady increase in ESP´ with depth was evident in all of the treatments. This is probably due to the high Na^+ levels present at the beginning of the study period, as the subsoil ESP´ decreased in relation to the baseline, except for the SLD at 60 cm and 150 cm, as well as the DLD at 150 cm soil depth. Deeper than 60 cm, the mean ESP´ of the SLD plots was higher than the DLD. This could be explained by the larger volumes of irrigation water applied to the latter plots, which facilitated the leaching of more Na^+ from the soil profile. The importance of leaching salts from the soil profile has been highlighted previously (Hussain, 1981). Rengasamy and Olsson (1993) predicted that Na^+ would accumulate in soil if the SAR of the applied water is greater than 3 and the leaching fraction is less than 50%. The high ESP´ observed in the subsoil of the SLD and DLD treatments remains a concern, as it may reduce the movement of water through the soil profile. Lower macro-porosity due to the accumulation of Na^+ and K^+ in the soil may affect the drainage capacity of soils, which in turn limits water percolation and, ultimately, the leaching of salts (Prior *et al*., 1992; Halliwell *et al*., 2001).

In soil with a permeable A horizon overlying a moderately draining B horizon, irrigation with treated municipal wastewater caused a reduction in *K* in the soil due to the increase of exchangeable Na^+ in the B horizon,

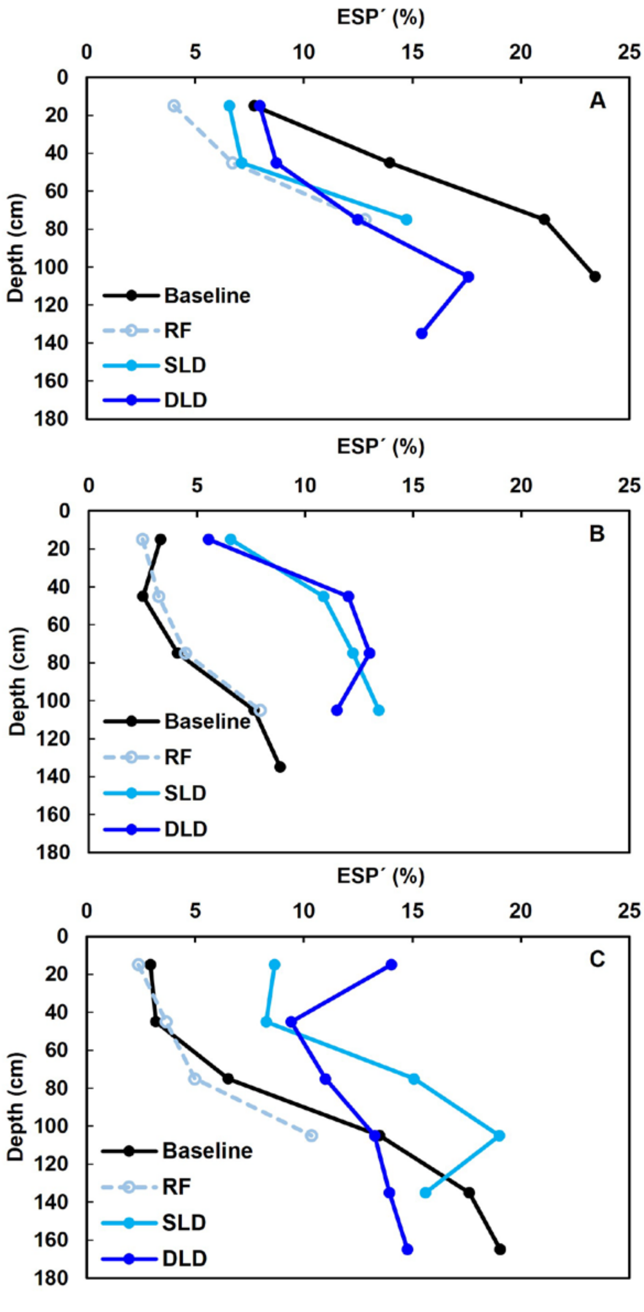

FIGURE 12
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil extractable sodium percentage (ESP´) of (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

which reduced the leaching of salts and led to increased soil salinity in the A horizon (Stevens *et al.*, 2003). Soil permeability problems will also occur if a solution with very low electrolyte concentration, such as rainwater, percolates through the soil (Du Plessis & Shainberg, 1985). Therefore, the application of large volumes of water with higher salinity, as is the case with the DLD treatments, might help to mitigate the accumulation of Na^+ at the soil surface and prevent reductions in *K* and IR.

Cl^-

The soil Cl^- content was not determined at the beginning of the study period; therefore, no baseline value is available. Soil Cl^- concentrations in the 0 cm to 30 cm layer increased with the amount of irrigation water applied to all the treatment plots (Fig. 14A to 14C). The topsoil Cl^- content of the shoulder site was 17 mg/kg, 35 mg/kg and 79 mg/kg for the RF, SLD and DLD plots, respectively (Fig. 14A). Soil Cl^- levels decreased sharply in the 30 cm to 60 cm layer of this landscape position and values were comparable between the three treatment plots. In the 60 cm to 90 cm soil layer, the Cl^- concentration of all the treatments increased again, suggesting that the subsoil has inherently high Cl^- levels. Similar results were observed in the backslope site, although the Cl^- content increased rather than decreased in the 30 cm to 60 cm soil layer (Fig. 14B). In the footslope site, the soil Cl^- content of the SLD and DLD plots were consistently higher than the RF plot and reached similar concentrations to those observed in the shoulder and backslope sites (Fig. 14C).

The mean Cl^- content of the topsoil increased with the amount of treated municipal wastewater applied (Fig. 15). The high Cl^- levels were to be expected, as the wastewater was disinfected by a chlorination treatment at the WWTW, resulting in a mean Cl^- content of 160 mg/L over the 12-year study period (Howell *et al.*, 2022). An accumulation of Cl^- seems evident at a depth of 90 cm, but cannot necessarily be ascribed to the irrigation water, as high Cl^- levels were observed in the subsoil of the RF treatments as well.

SOC

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

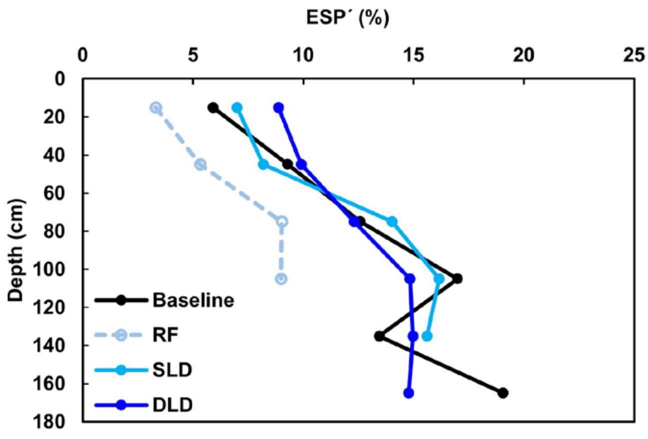

FIGURE 13
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean soil extractable sodium (ESP´) across the main experimental sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced.

Soil physical properties

Near-saturation hydraulic conductivity

No clear trend was observed that could explain the effect of irrigation with treated municipal wastewater on the K_{ns} at the shoulder site (Fig. 16A). However, the results were similar to what was reported by Walker and Lin (2008) for summit landscape positions irrigated with treated municipal wastewater for over 40 years. Similarly, there was little difference between the treatments at the footslope in terms of K_{ns}, despite the slightly lower K_{ns} measured in the DLD plot (Fig. 16C). These results were comparable to reports by Sparling *et al*. (2006) and Vogeler (2009), who observed no significant difference in K_{ns} between wastewater-irrigated and non-irrigated soils. In contrast, in the backslope site, K_{ns} decreased with an increase in the amount of treated municipal wastewater applied (Fig. 16B). The K_{ns} of the backslope site was 103 mm/h, 66 mm/h and 38 mm/h for the RF, SLD and DLD plots, respectively. Bedbabis *et al*. (2014) reported a significant decrease in the IR of sandy soil (5.5% clay) following four years of irrigation with treated municipal wastewater. The decrease was also significant in relation to a rainfed control treatment and one irrigated with well water (Bedbabis *et al*., 2014). In contrast, Lado and Ben-Hur (2010) reported improved saturated hydraulic conductivity (K_s) in sandy soil (12% clay) irrigated with secondary-treated municipal wastewater for more than 12 years.

A strong correlation was found between the EC_e of the 0 cm to 30 cm topsoil and the K_{ns} (Fig. 17). The relationship between EC_e and K_{ns} could be described best using a reciprocal-Y logarithmic-X model, where K_{ns} decreased significantly with an increase in EC_e up to an EC_e of 0.4 dS,/m whereafter it was expected to plateau. In contrast, Andrews *et al*. (2016) found no correlation between the K_s and EC_e of loam soils irrigated with treated municipal wastewater for over 50 years.

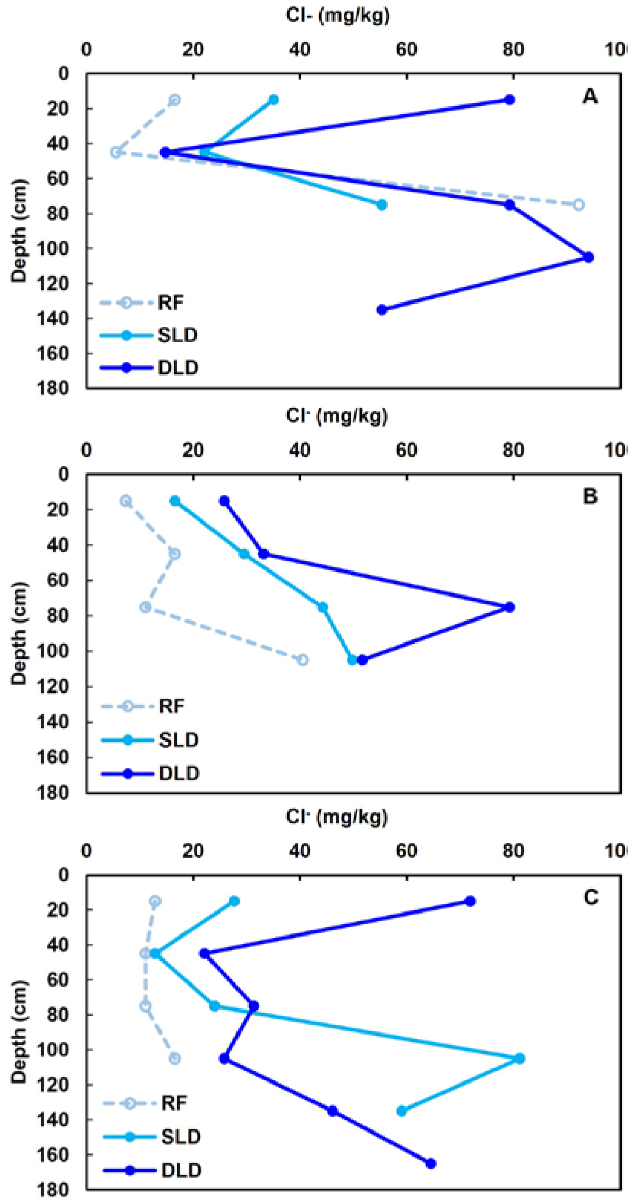

FIGURE 14
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the soil chloride content (Cl) of (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of irrigation with treated municipal wastewater.

CONCLUSIONS

Irrigation using treated municipal wastewater increased soil pH and EC_e. There also was an accumulation of Cl^- in the topsoil, which was likely due to the process of disinfection with chlorine at the wastewater treatment works. Substantial amounts of Na^+ and K^+ accumulated in the topsoil due to wastewater irrigation. Such soil K^+ increases could have a negative effect on the stability of wine colour should it be taken up by the grapevine in sufficient quantities, particularly if the levels of soil K^+ are such that it is absorbed excessively by the grapevines. In general, soil ESP´ increased as a result of irrigation with treated municipal wastewater. The increase was more prominent in the subsoil layers, possibly due to the seasonal leaching of salts by rainfall. Furthermore, the application of more water in the DLD treatment plots might also have contributed to more Na^+ being leached from the profile compared to the SLD plots. The K_{ns} at the surface of the soil could be related to the EC_e in the topsoil. Taking this into consideration, it may be beneficial in the future to quantify the formation of surface crusts that could form under wastewater irrigation. In addition, economically viable practices should be developed to alleviate such surface crusts. It should be noted that the results of this study represent specific in-field situations in three commercial vineyards under one set of climatic conditions. Future research should focus on the use of treated municipal wastewater for irrigation of vineyards or other crops on different soil types in different climatic regions.

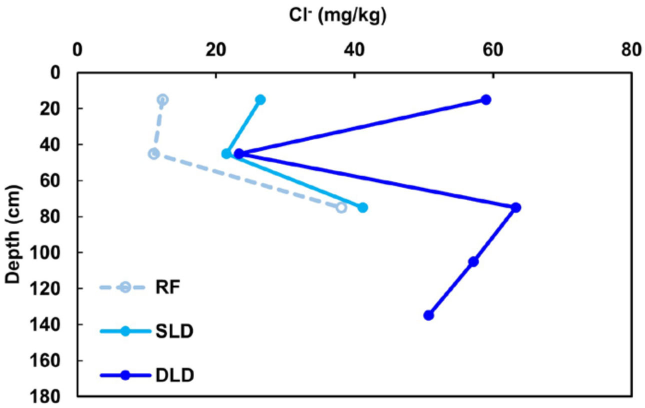

FIGURE 15
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the mean soil chloride (Cl) content across the main experimental sites after 11 years of irrigation with treated municipal wastewater.

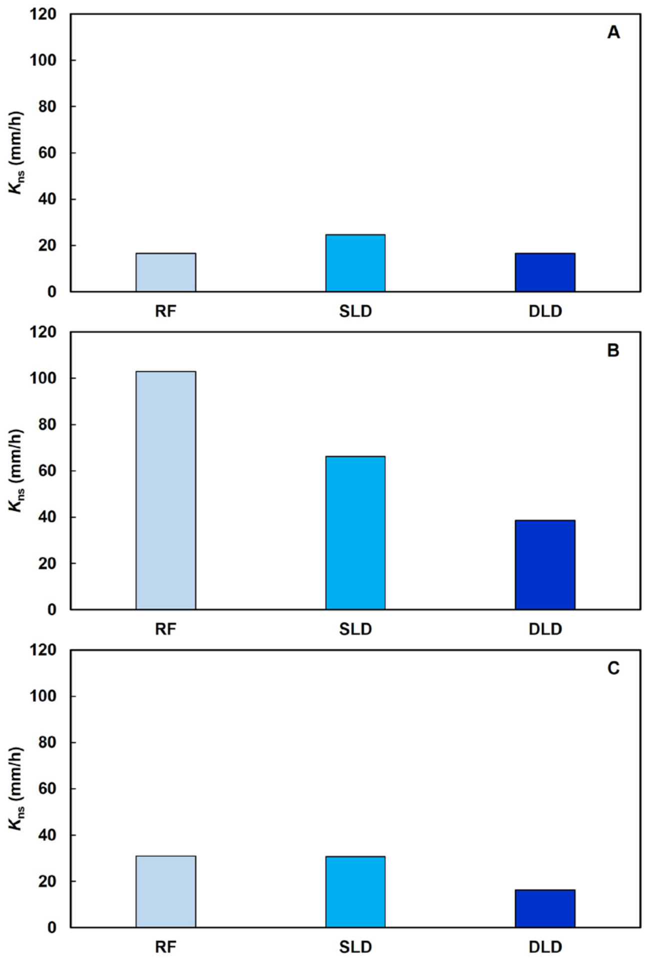

FIGURE 16
Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double dripper line (DLD) on the near-saturation hydraulic conductivity (K_{ns}) (suction = 2 cm) of (A) a shoulder, (B) a backslope and (C) a footslope during the 2017/2018 season.

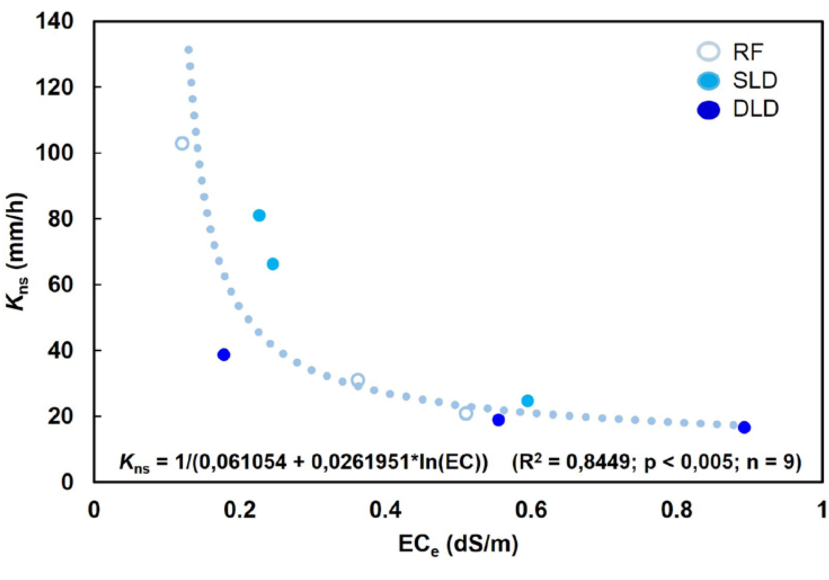

FIGURE 17
Effect of electrical conductivity (EC_e) of the 0 cm to 30 cm topsoil layer on near-saturation hydraulic conductivity (K_{ns}) during the 2017/2018 season.

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