

Long-term impacts of pasture irrigation with treated sewage effluent on nutrient status of a sandy soil in Zimbabwe

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Abstract Declining freshwater resources and the need to safely dispose wastewater have led to a rapid increase in wastewater reuse in developing countries. However, empirical evidence on the effects of effluent-irrigation on soil fertility is limited. The study investigated the nutrient status of a sandy soil after 26 years of effluent irrigation. Soil samples from effluent-irrigated and non-irrigated sites were analysed for pH, electrical conductivity (EC), soil organic carbon (SOC), total and plant available forms of N and P, exchangeable bases and trace metals. Analysis of effluent quality showed that, besides Cr and Cd, all measured parameters were within acceptable limits for wastewater irrigation. Our results revealed that effluent-irrigation significantly ($P < 0.05$) enriched the soil with essential nutrients

for plant growth, which are commonly deficient in most soils of Zimbabwe. Effluent-irrigated soils had significantly ($P < 0.05$) higher pH, EC, SOC, total and available N and P and, exchangeable Ca and Mg at 0–30 cm-depth. However, apart from Cr accumulation, effluent irrigation significantly ($P < 0.05$) depleted Zn, Cu and Cd probably due to plant uptake and enhanced mobility under acidic soil pH. Cr accumulation and depletion and mobility of Zn, Cu and Cd in effluent-irrigated soils could threaten the sustainability of the practice. We recommend a review of the current management practices based on crop water requirements, effluent quality and environmental considerations.

Keywords Effluent irrigation · Nutrients · Pasture · Wastewater

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Introduction

Increasing demand for freshwater resources has led to the use of poor quality water such as industrial and domestic wastewater for irrigation. Besides being relatively cheaper than freshwater, wastewater is a valuable source of plant nutrients and organic matter vital for maintenance of soil fertility and productivity (Rattan et al. 2005). The nutrient content of wastewater is highly variable, depending on source and method of treatment. Moreover, wastewater irrigation is considered an environmentally safer disposal

option than direct discharge into surface water bodies (Khouri et al. 1994). However, wastewater also contains variable concentrations of toxic trace elements and pathogenic organisms. The need to understand environmental impacts of wastewater irrigation has led to a proliferation of studies encompassing wastewater quality (e.g. Papadopoulos 1985; Pescod 1992), and subsequent effects on crop productivity (e.g. Khouri et al. 1994; Oloya and Tagwira 1996b) and soil quality (e.g. Nyamangara and Mzezewa 1999, 2000; Rattan et al. 2005).

Wastewater contains 99% water and adequate N, P and K to meet crop requirements (Mara and Cairncross 1984). High crop yields have been reported following wastewater irrigation (e.g. Khouri et al. 1994; Oloya and Tagwira 1996b). Khouri et al. (1994) observed yield increases of between 28 and 51% in wheat, mung beans, rice, cotton and potatoes using wastewater, compared to freshwater irrigation plus commercial fertilisers. In Zimbabwe, increased wheat and soyabean yields were reported using a mixture of sewage effluent and sludge (Oloya and Tagwira 1996b). These findings have been attributed to the nutrient content of the effluent as well as improved soil physical properties due to presence of organic matter. In Jordan, Mohammad and Mazahreh (2003) observed increased soil P, K, Fe, Mn and soil organic carbon (SOC) levels in wastewater-irrigated soils compared to those irrigated with freshwater. Rattan et al. (2005) reported a 59% increase in soil organic carbon and a drop in soil pH of 0.4 units after 20 years of wastewater irrigation. In Zimbabwe, Nyamangara and Mzezewa (1999) reported a rise in soil pH following application of a mixture of sewage effluent and sludge whereas Mapanda et al. (2005)'s findings ranged from no effects to significant pH increases. Both soil pH and SOC influence nutrient and trace elements behaviour in soils. However, the effects of wastewater irrigation on both soil fertility and crop yields appear to vary depending on soils, element under consideration, wastewater quality, crop species and climatic conditions.

In Zimbabwe, limited studies have been conducted to investigate the long-term effects of effluent irrigation on soil fertility. To date, most of the studies have focused on trace metal accumulation in soil and plants after the application of either sludge or a mixture of sludge and sewage effluent (Oloya and

Tagwira 1996a, b; Nyamangara and Mzezewa 1999; Mapanda et al. 2005). These studies showed elevated levels of selected trace metals on both light and heavy textured soils. For example, Oloya and Tagwira (1996a, b) observed high levels of Zn, Pb, Cu and Ni on a sandy soil irrigated with a mixture of effluent and sludge. However, higher Zn and Cd levels were reported in virgin soils than the wastewater-irrigated soils. They attributed their observations to Cd and Zn mobility and leaching by irrigation water. Similarly, unexplained high levels of Cr were reported in the control compared to wastewater-irrigated sandy soils (Mapanda et al. 2005). In contrast, trace metal accumulation in topsoil were reported by Nyamangara and Mzezewa (1999) on a clayey soil following application of sludge and effluent. No trace metal mobility and leaching were reported probably due to high adsorption on such soils. In Jordan, wastewater irrigation lowered soil pH, but significantly increased soil salinity, Fe, Mn and soil organic carbon levels (Mohammad and Mazahreh 2003). No effects on Zn and Cu concentrations were reported. However, most of these studies have been limited to sites that have received wastewater irrigation for <10 years (e.g. Mohammad and Mazahreh 2003; Mapanda et al. 2005). As such, these studies failed to resolve long-term impacts of wastewater irrigation, particularly on plant nutrients. Literature cited from Zimbabwe was based on studies conducted on sites located close to wastewater treatment plants. Consequently, mixtures of sludge and effluent were applied in all cases (e.g. Oloya and Tagwira 1996a, b; Nyamangara and Mzezewa 1999, 2000; Mapanda et al. 2005). As such, the long-term effects of continuous irrigation using treated sewage effluent without sludge application have not been investigated in Zimbabwe. Compared to sludge, sewage effluent has low concentrations of both nutrient and trace elements (Ngole et al. 2006). However, when effluent is used continuously as the sole source of irrigation water, nutrient and trace elements may accumulate to phytotoxic levels and adversely affect soil productivity. The objective of this study was to investigate nutrient status of a sandy soil after 26 years of continuous pasture irrigation with treated sewage effluent. Contrary to previous investigations, the study site was located about 50 km from the sewage treatment plant, and no sludge had been applied due to long haulage distance.

Materials and methods

Description of study site

The study was carried out at a site (30°55' E, 18°05' N) in Beatrice about 22 km from the capital city, Harare. The site experiences sub-tropical climate with mean annual rainfall of 850 mm and mean annual temperature of 23°C. Rainfall is restricted to a well-defined wet season which extends from late November to March. Soils were predominantly deep (>1 m) sands and loamy sands derived from granite, with distinct clay eluviation. They were classified as Harare 6G.3 according to the Zimbabwe classification, *Gleyic Luxisol* according to FAO soil classification and *Udic kandiuustalf* according to USDA soil classification (Nyamapfene 1991). A typical profile described by Purves (1976) at the site had loamy sandy texture at 0–60 cm, overlying coarse sand at 61–100 cm and coarse sandy clay with gravel at 101–120 cm and below. The soils have high potassium (164 mg kg⁻¹) while N and P are generally deficient. They also have low phosphate sorption capacity and acidic pH (≈4.3) arising from excessive leaching of bases (Nyamapfene 1991). Physico-chemical characteristics of the soils were described by Purves (1976).

The study site had a total of 1444 ha of which about 481 ha had been under continuous effluent irrigation since 1979. Pasture consisted of a mixture of Kikuyu (*Pennisetum clandestinum*, Chiov.) and star grass (*Cynodon dactylon*, L.) under sprinkler and furrow irrigation, and an additional 80 ha of bana grass (*Pennisetum typhoides*, L.) under furrow irrigation. Effluent for irrigation was conveyed by an asbestos–cement pipeline from the trickling filters at sewage treatment plant about 50 km from the site. The plant treated wastewater from a nearby city with an estimated population of 300,000 people. At the study site, effluent was stored in six maturation ponds operating in series in winter and in parallel in summer. The maturation ponds consisted of 5 aerobic ponds of width and breadth of 75 m each and depth of 2 m, and an anaerobic pond of width and breadth of 150 m each and depth of 5 m. The ponds were designed with a total mean residence time of 9 days, but increased wastewater generation due population growth has led to system overloading. The objective of effluent irrigation was to avoid direct discharge

into surface water bodies rather than planned irrigation based on crop water requirements. As such, the amount and frequency of irrigation were meant to avoid effluent accumulation and overflowing of maturation ponds. At the time of the study, the ponds received 36 Ml day⁻¹ of wastewater against a design flow of 21750 m³ day⁻¹. Historical data from farm records showed that total Kjeldahl nitrogen ranged between 30 and 40 mg l⁻¹ in summer and 60 mg l⁻¹ in winter. The irrigation system was operated on a 12-h cycle with a 3D interval. During the rainy season, excess effluent was discharged through runoff channels into natural grassland. Values from farm records and estimates from pump discharges gave effluent application rates of 15 Ml ha⁻¹ yr⁻¹ at commencement of effluent-irrigation and 36.5 Ml ha⁻¹ yr⁻¹ at the time of the study, corresponding to 1500 and 3650 mm ha⁻¹ yr⁻¹, respectively. When combined with annual rainfall of 850 mm, this gave a total hydraulic loading rate of 2300 and 4500 mm ha⁻¹ yr⁻¹, respectively. At the commencement of irrigation, 2300 mm ha⁻¹ yr⁻¹ was considered to be more than double the water requirement of the pasture.

Effluent sampling and analysis

Effluent samples were collected in sterile 2-l plastic bottles every 20 min from 0900 to 1600 h from the main canal conveying effluent from the maturation ponds to the fields. Samples were collected monthly in summer from January to March and refrigerated at 4°C until analysis, which was done within 24 h in all cases. Effluent pH and electrical conductivity (EC) were determined in the field using pH and EC meters (Model: ELE Paqualab, ELE International Ltd 1996), respectively. Standards methods for wastewater analysis (APHA 1989) were used to analyse effluent samples. Total nitrogen (TN), total phosphate (TP), Zn, Cr, Cd and Cu were determined on unfiltered samples while NH₄-N, NO₃-N, Mg, Ca, K and Na were determined on samples filtered through a 0.45-µm filter paper following the method of Chapman (1998). TN was determined by the Kjeldahl method. TP was determined by the ascorbic acid method following digestion with HClO₄–HNO₃ solution. Zn, Cu, Cd and Cr were determined by atomic absorption spectrophotometry after double acid (HNO₃–HClO₄) digestion. NH₄-N was determined by direct nesslerisation method of APHA (1989)

followed by photometric determination. Dissolved phosphate was determined using the vanadate–molybdate method. $\text{NO}_3\text{-N}$ was determined by the Palintest method (ELE International Ltd 1996). Effluent data were evaluated using the FAO and Zimbabwe guidelines for assessing the suitability of wastewater for irrigation. The Zimbabwe system only specifies limits for trace metals while the FAO system include both trace metals and other chemical parameters (SAR, pH, EC, $\text{NO}_3\text{-N}$, B, Cl^- , Na) with potential impacts on salinity, infiltration and specific ion toxicity (Papadopoulos 1985; Zimbabwe Water Regulations 2000). SAR was calculated according to the FAO system:

$$\text{SAR} = [\text{Na}^+]/(\sqrt{([\text{Ca}^{2+}]/2 + [\text{Mg}^{2+}]/2)})$$

where [] denotes concentration in meq l^{-1} .

Soil sampling and analysis

Three sites with similar soils were identified for sampling within the study area. Two blocks grown to a mixture of *P. clandestinum* and *C. dactylon* that have been continuously effluent-irrigated for 26 years were selected. One sprinkler-irrigated block, adjacent to the maturation ponds, was the most frequently irrigated. The other block was furrow-irrigated and about 1 km from the ponds. Due to its distance from the maturation ponds it received erratic application of effluent. A non-irrigated site dominated by *Hyperhemia* species and about 5 km upgradient of the effluent-irrigated area was used as a control. Four replicates of soil samples were collected from each block at 30 cm-depth intervals up to 90 cm using a 50 mm diameter steel hand auger. During sampling, areas that showed evidence of localised ponding, cow dung, erosion and other visible disturbances were avoided. To avoid contamination, sampling started from the control to the irrigated sites. Samples were collected in clean polythene bags, labelled and transported to the laboratory in a cooler box.

Samples were air-dried in a well-ventilated room in wooden boxes following recommendations of Page et al. (1982). They were then ground using a mortar and pestle and sieved through a 2 mm diameter steel sieve. Samples were stored at 4°C until analysis, which in all cases was done within 72 h. Recommended standard methods of soil analysis were used (Page et al. 1982). Soil pH and EC were determined potentiometrically in 1:1 suspension (soil: 0.01 CaCl_2

for pH and soil: water for EC) using a pH and EC meter. Total N was determined using the Kjeldahl procedure. $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ were determined by the colorimetric procedure after KCl extraction. Soil organic carbon (OC) was determined by the modified Walkley–Black method of Houba et al. (1989) with additional heating supplied by reflux. Total P was determined by the ascorbic acid method after $\text{HClO}_4\text{-HNO}_3$ acid digestion. Bray-1 P was extracted by Bray-1 method and determined as for total P. Exchangeable cations were extracted by acidified ammonium acetate followed by atomic absorption spectrophotometric determination for Ca and Mg, and flame emission for K and Na. Zn, Cu, Cr and Cd were extracted using the USEPA double acid digestion method which recovers almost 100% of the metals from the sample (Henning et al. 2001). Samples were weighed, ashed and taken up in $\text{HClO}_4\text{-HNO}_3$ solution and made up to a known volume. The concentrations of Zn, Cu, Cd and Cr in the soil extract were determined with an atomic absorption spectrophotometer. Zn, Cu, Cd and Cr concentrations were compared to FAO guideline limits for wastewater-irrigated soils (Pescod 1992). Currently, Zimbabwe has no locally derived guideline limits for heavy metal concentrations in wastewater-irrigated soils. Soil data were tested for effects of effluent irrigation by analysis of variance (ANOVA) using GENSTAT 5 statistical package (Rothamsted Experiment Station 1995). Treatment effects were considered significant at 5% probability level. Data that lacked normality were log-transformed before ANOVA.

Results and discussion

Effluent chemical characteristics

Table 1 shows chemical characteristics of the effluent used for irrigation. The effluent used for irrigation was almost neutral with a pH value of 7.2 ± 0.07 and non-saline with an average EC of $0.98 \pm 0.036 \text{ dS m}^{-1}$. The values are within acceptable ranges according to FAO wastewater reuse guidelines. The EC values will have negligible salinity effects. The effluent had higher TN ($35.1 \pm 0.42 \text{ mg l}^{-1}$) and $\text{NH}_4\text{-N}$ ($13.7 \pm 1.29 \text{ mg l}^{-1}$) than $\text{NO}_3\text{-N}$ ($1.0 \pm 0.15 \text{ mg l}^{-1}$) indicating inefficient wastewater treatment and incomplete nitrification probably due to overloading of the

Table 1 Chemical characteristics of effluent used for irrigation and guideline limits for unrestricted long-term irrigation using wastewater

Parameter	Month of sampling			Mean \pm std. deviation	Guideline limit for long-term irrigation	
	January	February	March		FAO ^a	Zimbabwe ^b
TN (mg l ⁻¹)	35.6	34.8	35.0	35.1 \pm 0.42	–	–
NH ₄ -N (mg l ⁻¹)	13.2	12.8	15.2	13.7 \pm 1.29	–	–
NO ₃ -N (mg l ⁻¹)	0.9	1.2	1.0	1.0 \pm 0.15	5	–
DP (mg l ⁻¹)	3.0	3.0	2.8	2.9 \pm 0.12	–	–
TP (mg l ⁻¹)	7.2	8.1	9.1	8.1 \pm 0.36	–	–
K (mg l ⁻¹)	29.7	30.4	29.9	30.0 \pm 0.36	–	–
Na (mg l ⁻¹) ^c	46.0 (2.0)	46.6 (2.0)	47.7 (2.1)	46.0 \pm 0.86 (2.0 \pm 0.04)	72.0 (3.0)	–
Mg (mg l ⁻¹) ^c	8.9 (0.7)	9.7 (0.8)	11.5 (1.0)	10.0 \pm 0.45 (0.8 \pm 0.04)	–	–
Ca (mg l ⁻¹) ^c	22.2 (1.1)	26.6 (1.3)	23.1 (1.2)	24.0 \pm 2.32 (1.2 \pm 0.12)	–	–
pH	7.2	7.2	7.1	7.2 \pm 0.07	6.5–8.4	–
EC (dS m ⁻¹)	0.95	1.02	0.98	0.98 \pm 0.036	2.0	–
SAR ((meq l ⁻¹) ^{1/2}) ^d	2.1	1.9	1.9	2.0 \pm 0.02	3.0	–
Zn (mg l ⁻¹)	0.06	0.05	0.06	0.06 \pm 0.017	2.0	–
Cu (mg l ⁻¹)	0.02	0.01	0.01	0.01 \pm 0.010	0.2	0.2
Cd (mg l ⁻¹)	0.03	0.05	0.02	0.03 \pm 0.017	0.01	0.01
Cr (mg l ⁻¹)	1.42	1.48	1.31	1.40 \pm 0.011	0.1	0.1

DP: Dissolved P; TP: Total P, TN: Total N

Sources: ^a Pescod 1992; ^b Zimbabwe Water Regulation 2000; ^c Figures in parentheses show concentration in meq l⁻¹;
^d SAR = [Na⁺]/(√([Ca²⁺]/2 + [Mg²⁺]/2))

treatment facilities. Similar values were reported at the commencement of irrigation and subsequent studies by Bengtsson et al. (1998). However, the NO₃-N concentrations observed were lower than those reported by Emongor and Ramolemana (2004) in effluent in Botswana (6.7–29.3 mg l⁻¹). Average total and dissolved P concentrations were 8.1 \pm 0.36 and 2.9 \pm 0.12 mg l⁻¹, respectively. Low dissolved P in effluent is due to its inherent low solubility in water and its tendency to undergo adsorption (Vaisman et al. 1981; Gallegos et al. 1999). Basic cation concentrations (mg l⁻¹) increased in the order: Na > K > Ca > Mg with average concentrations of 46.0 \pm 0.86, 30.0 \pm 0.36, 24.0 \pm 2.32 and 10.0 \pm 0.45. High K concentration observed was in contrast to Todd (1980) who reported low values in effluent. This source of the high K could be due to contamination during the conveyance of the effluent. Compared to other basic cations, Na was observed in highest concentration. Sources of Na in effluent include soaps, detergents and other industrial chemicals. Calcium concentration in the effluent averaged 24 mg l⁻¹ and possible sources include asbestos-cement pipes used to convey sewage.

Although no limits exist for the individual cations, a computed SAR value of 2.0 \pm 0.02 was lower than the FAO limit of 3. SAR is a measure of the potential of irrigation water to cause permeability problems. Average concentrations of Zn, Cu, Cd and Cr were 0.06 \pm 0.017, 0.01 \pm 0.010, 0.03 \pm 0.017 and 1.40 \pm 0.011. Concentrations of Zn and Cu remained lower than the recommended maximum levels in irrigation water while Cd and Cr exceeded the limits. Sources of trace metals in municipal effluent include electroplating industries for Zn and Cr, and batteries for Cd and Cu. All trace metal concentrations were about 10-fold lower than those from industrialised countries as reported in Oloya and Tagwira (1996a). However, heavy metals have a tendency to accumulate in soil and plants with continuous wastewater irrigation. Therefore, periodic monitoring of soil and plants is required.

Nutrient loading rates

Table 2 shows annual loading rates (ALR) for selected nutrient elements in effluent and typical

Table 2 Annual loading rates (ALR), pasture requirements and UK guideline limits for selected macro- and micronutrients

Parameter	ALR (kg ha ⁻¹ yr ⁻¹)	
<i>Macronutrients</i>	<i>Pasture requirement</i>	
TN	1,281	350–600 ^a
NH ₄ -N	106	–
NO ₃ -N	296	–
TP	500	30–40 ^a
DP	37	–
K	1,095	225–340 ^a
<i>Micronutrients</i>	<i>UK guideline limit (kg ha⁻¹ yr⁻¹)</i>	
Mg	365	–
Ca	876	–
Zn	2.2	15 ^b
Cu	0.4	7.5 ^b

Sources: ^a Tchobagolous 1991; ^b Mapanda et al. 2005

TN: total nitrogen; TP: total P; DP: dissolved P

pasture requirements. ALR for TN, NH₄-N and NO₃-N were 1281, 106 and 296 kg ha⁻¹ yr⁻¹, respectively, compared to an annual pasture requirement of 350–600 kg N ha⁻¹ yr⁻¹. Average ALR for TP (500 kg ha⁻¹ yr⁻¹) was more than 12-fold the annual pasture requirement (30–40 kg P ha⁻¹ yr⁻¹) while that of DP (37 kg ha⁻¹ yr⁻¹) was adequate. Average ALR for K (1095 kg ha⁻¹ yr⁻¹) was about 3-fold higher than pasture requirement (200–290 kg ha⁻¹ yr⁻¹) while those for Mg and Ca were 365 and 876 kg ha⁻¹ yr⁻¹, respectively. Zn and Cu had the lowest loading rates, averaging 2.2 and 0.4 kg ha⁻¹ yr⁻¹, respectively. In comparison, the annual N, P, K, Mg, Ca, Zn and Cu exceeded the general recommendation for most agronomic crops grown in Zimbabwe (e.g. maize, wheat, cotton). For example, a basal dressing of 250 kg ha⁻¹ Compound D (6:14:7%; N:P:K) and a top dressing of 250 kg ha⁻¹ ammonium nitrate (34.5% N) are commonly used in Zimbabwe. Mg, Ca, Zn and Cu are rarely included in fertiliser formulations except for specific crop species and in cases where acute deficiencies have been reported. However, whilst Zimbabwe has no guideline limits for Zn and Cu loading rates, comparison with UK limits showed that both elements were within acceptable limits. High Cu and Zn concentration in effluent has potential to cause phytotoxicity to sensitive crops especially under acidic conditions. The results clearly showed that effluent had adequate

macronutrients (N, P, & K), Mg, Ca, Zn and Cu. Consequently, no deficiency symptoms were evident in pasture at the study site. Our findings concur with yield data reported by Khouri et al. (1994) and Oloya and Tagwira (1996b). They reported significantly higher yields under wastewater irrigation in wheat, soyabeans, mung beans, rice, cotton and potatoes.

Effects of effluent irrigation on soil fertility

Soil pH and electrical conductivity

Across treatments, highly acidic soil pH values were observed ranging between 3.8 and 5.4 (Table 3). Acidity is a major soil fertility problem on sandy soils in high rainfall areas (850–1,200 mm yr⁻¹) of Zimbabwe (Grant 1995). Under such conditions, soil acidity is caused by excessive leaching of basic cations as reported by Nyamapfene (1991). On such soils, lime application is required for most agronomic crops. Comparable soil pH values have been observed on similar soils in Zimbabwe, where it has been reported to induce P fixation and Al toxicity (Nyamangara et al. 2000). However, effluent-irrigated soils had 23–42% ($P < 0.05$) higher pH than control soils due to the neutralising effect of effluent with near-neutral pH (7.2). Similar effects on soil pH have been reported in Zimbabwe after application of sewage sludge and effluent (Nyamangara and Mzezewa 1999). Mapanda et al. (2005) reported a pH rise of between 0.5 and 3 units, while by contrast Rattan et al. (2005) observed a pH drop of 0.4 units after wastewater irrigation.

Average EC values (dS m⁻¹) varied from 0.26 to 0.30 for the control, 0.65 to 0.70 for furrow-irrigated soils and 0.63 to 0.68 for sprinkler irrigated-soils. Effluent irrigation significantly ($P < 0.05$) increased EC values by an average of between 126 and 154% over the control. A gradual decline in EC with depth, though not significant was also observed in all treatments, probably due to evaporation effect. The EC increase in effluent-irrigated soils was attributed to salt-loading caused by continuous irrigation with effluent. Similar increases have been reported by Mohammad and Mazahreh (2003) following irrigation with secondary treated municipal wastewater. They also observed that lower wastewater application rates equivalent to pan evaporation increased salinity more in the topsoil while a higher rate (equivalent to

Table 3 Average pH, EC (dS m⁻¹) and concentrations (mg kg⁻¹ unless specified) of OC, total and available N and P in effluent-irrigated and non-irrigated control soils

Site	pH	EC (dS m ⁻¹)	OC (g kg ⁻¹)	TN (g kg ⁻¹)	TP	Bray-1 P	NH ₄ -N	NO ₃ -N
0–30 cm								
Control	3.9 a	0.30 a	7.1 a	0.4 a	54.3 a	15.3 a	2.3 a	1.5 a
Furrow	4.9 a	0.70 b	15.8 b	1.4 b	241.5 b	78.1 b	3.0 b	5.8 b
Sprinkler	5.0 a	0.68 b	17.2 b	1.6 b	274.5 b	69.5 b	3.7 c	2.3 c
Sig. (<i>P</i> = 0.05) ^a	N.S	*	*	*	*	*	*	*
30–60 cm								
Control	3.9 a	0.27 a	3.8 a	0.2 a	45.8 a	8.5 a	1.9 a	1.8
Furrow	4.8 a	0.65 b	4.2 a	0.3 a	46.5 a	25.8 b	2.0 a	0.9
Sprinkler	5.0 a	0.67 b	4.3 a	0.2 a	50.1 a	28.4 b	2.3 a	1.0
Sig. (<i>P</i> = 0.05)	N.S	*	N.S	N.S	N.S	*	N.S	*
60–90 cm								
Control	3.8 a	0.26 a	2.9 a	0.1 a	47.2 a	5.0 a	1.2 a	1.7 a
Furrow	5.0 a	0.65 b	3.1 a	0.2 a	48.3 a	30.9 b	1.7 a	0.3 b
Sprinkler	5.4 a	0.63 b	3.3 a	0.2 a	47.7 a	32.3 b	2.5 b	0.4 b
Sig. (<i>P</i> = 0.05)	N.S	*	N.S	N.S	N.S	*	*	*

TP: Total P; TN: Total N; OC: Soil organic carbon

^a N.S: Not significant and *: significant at *P* = 0.05, respectively

^b Means in the same column followed by different letters are significantly different at *P* = 0.05

125% of pan evaporation) had the same effect in the subsoil. However, all EC values were lower than the 2 dS m⁻¹ limit, and therefore negligible salinity effects on germination and plant growth were expected from effluent-irrigation.

Soil organic carbon

Average soil organic carbon levels (g kg⁻¹) at 0–30 cm-depth were 15.8 for furrow-irrigated and 17.2 for sprinkler-irrigated soils, corresponding to more than 100% increase over the control (7.1 g kg⁻¹). The increases were only significant (*P* < 0.05) at 0–30 cm-depth but not 30–60 and 60–90 cm-depths, where average levels across treatments were 3.8–4.3 and 2.9–3.3 g kg⁻¹, respectively (Table 3). Rattan et al. (2005) reported average SOC levels equivalent to 6.5 g kg⁻¹, corresponding to a 59% increase over the control in wastewater-irrigated sandy soils. Both their SOC levels and per cent increases were lower than observations made in this study probably due to differences in effluent quality and application rates. The build-up of SOC in effluent-irrigated soils shows an improvement in soil quality, with implications on soil fertility and productivity. Soil organic carbon

improves soil structural properties, acts as a nutrient reserve and counteracts the effects of salinity (Rattan et al. 2005). Organic matter also sequesters trace metals and reduces their mobility and risk of leaching. However, it was unclear whether the increased SOC in effluent-irrigated soils originated from effluent or decomposing biomass from pasture.

Total phosphate and Bray-1 phosphate

Effluent-irrigation significantly (*P* < 0.05) increased both total phosphate and Bray-1 P compared to the control site, which had typical concentrations for such soils as observed by Grant (1995). Across soils depths, average TP concentrations (mg l⁻¹) were 47.2–54.3, 46.5–241.5 and 47.7–274.5 for control, furrow-irrigated and sprinkler-irrigated soils, respectively. In effluent-irrigated soils TP significantly (*P* < 0.05) declined with depth, dropping to levels similar to that of the control at 30–60 and 60–90 cm-depths. Bray-1 P showed the same trend as TP but effluent-irrigated soils had significantly (*P* < 0.05) higher concentration than the control at all depths. Across soil depth average Bray-1 P concentrations (mg l⁻¹) were between 5.0–15.3, 25.8–78.1 and

28.4–69.5 for control, furrow-irrigated and sprinkler-irrigated soils. Both TP and Bray-1 P were highest at 0–30 cm-depth suggesting adsorption of P applied with effluent. However, significantly high ($P < 0.05$) Bray-1 P concentrations at 30–60 and 60–90 cm-depths in effluent-irrigated soils compared to the control indicated P mobility. P is generally considered immobile, but its mobility has been reported on natural and wastewater-irrigated sandy soils of low P sorption capacity (Nyamapfene 1991). In addition, Withers and Sharpley (1995) estimated that significant P leaching might occur after 25% of the soil P sorption capacity has been saturated. The mobility could have been enhanced by over-irrigation. Despite the mobility, P concentrations in effluent-irrigated soils were considered adequate for most crops. This was in agreement with claims by Mara and Cairncross (1989) who reported that wastewater had adequate P to meet crop requirements.

Total nitrogen, ammonium and nitrate

Effluent-irrigated sites had significantly ($P < 0.05$) higher TN than control site at 0–30 cm-depth, but

similar at the other two depths. This increase in TN in the topsoil was due to nitrogen loading through effluent irrigation. Consequently, significantly ($P < 0.05$) higher $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentrations were observed in the top 30 cm-depths of effluent-irrigated soils than control. Below 30 cm-depth both $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ showed no consistent trend probably due to a variety of processes that influence N dynamics in the soil. Gallegos et al. (1999) attributed this to the diversity of microbial processes that drives the nitrogen cycle. Both $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentrations in effluent-irrigated soils were very low ($0.3\text{--}5.8 \text{ mg l}^{-1}$) considering the total N loading from wastewater. Deep-rooted pasture grasses are efficient in N removal from soils. For example, Rhodes grass (*Chloris gayana*) had been reported to extract and accumulate nitrates and prevent deep seepage and groundwater pollution (Vaisman et al. 1981). Sprinkler-irrigated soils had higher $\text{NH}_4\text{-N}$ than $\text{NO}_3\text{-N}$ at all depths. This was probably due to incomplete nitrification caused by prolonged anaerobic conditions arising from frequent irrigation.

Table 4 Concentration (mg kg^{-1}) of exchangeable bases and selected trace metals in effluent-irrigated and non-irrigated control soils, and FAO limits for wastewater-irrigated soils

Site	K	Ca	Mg	Na	Cd	Zn	Cr	Cu
0–30 cm								
Control	119.0 a	56.0 a	7.1 a	3.3 a	6.0 a	50.5 a	12.0 a	22.8 a
Furrow	120.2 a	66.5 b	10.9 b	3.6 a	5.5 a	32.8 b	17.8 b	17.1 b
Sprinkler	122.1 a	70.0 b	11.3 b	3.5 a	4.9 b	38.5 b	13.5 a	15.7 b
Sig. ($P = 0.05$)	N.S	*	*	N.S	*	*	*	*
30–60 cm								
Control	82.7 a	54.8 a	5.2 a	3.0 a	5.8 a	48.8 a	11.8 a	8.3 a
Furrow	90.3 a	58.9 a	5.7 a	3.2 a	6.5 a	34.8 b	18.3 b	25.8 b
Sprinkler	87.0 a	59.5 a	5.4 a	3.1 a	4.8 b	46.3 a	13.8 a	27.2 b
Sig. ($P = 0.05$)	N.S	N.S	N.S	N.S	*	*	*	*
60–90 cm								
Control	80.9 a	43.9 a	4.5 a	3.2 a	6.5 a	61.5 a	12.0 ^a	7.3 ^a
Furrow	76.4 a	44.7 a	4.6 a	3.4 a	6.8 a	53.8 a	16.8 ^b	15.1 ^b
Sprinkler	74.3 a	45.0 a	4.7 a	3.3 a	5.8 a	63.5 a	13.5 ^a	17.5 ^b
Sig. ($P = 0.05$) ^a	N.S	N.S	N.S	N.S	N.S	N.S	*	*
FAO limits (mg kg^{-1})	–	–	–	–	1–3	150–300	N/A ^b	300

Means in the same column followed by different letters are significantly different at $P = 0.05$

^a N.S: Not significant and *: significant at $P = 0.05$, respectively

^b N/A: No limit exists for Cr for wastewater-irrigated soils

Exchangeable bases

The top 60 cm of the irrigated sites had higher K (1–9%) than the control although differences were not significant (Table 4). The levels of K were high and comparable at all sites. Such values are expected on sandy soils of granitic origin due to high inherent K in parent material. Consequently, effluent irrigation had no significant effect on K concentration. Sandy soils in Zimbabwe have sufficient K; hence no K deficiencies have been reported on such soils (Nyamapfene 1991; Nyamangara et al. 2000). Therefore the high native K could have masked the effects of K added through irrigation. Moreover, K is required by plants in high amounts and is prone to luxury consumption (Landon 1991). Significantly ($P < 0.05$) higher concentrations of Mg (19–25%) and Ca (54–59%) were observed in irrigated sites than the control at 0–30 cm-depth. Concentrations were not significantly different at lower depths, probably because Ca and Mg are held at the soil exchange sites. Ca and Mg also showed high variability within treatments (coefficient of variation $>20\%$). Effluent irrigation had no significant effect on Na concentrations, which ranged between 3.0 and 3.6 mg l⁻¹. Per cent Na increases due to effluent-irrigation were between 3 and 9, with highest increase in the 0–30 cm-depth.

Cadmium, zinc, copper and chromium

Table 4 shows the average concentration of Zn, Cu, Cd and Cr in effluent-irrigated soils and control. Cadmium concentrations exceeded the limit of 3 mg kg⁻¹, with average concentrations ranges of 6.0–6.5, 5.5–6.8 and 4.8–5.8 mg kg⁻¹ for the control, furrow and sprinkler-irrigated sites, respectively. Effluent-irrigation significantly ($P < 0.05$) depleted soil Cd at 0–30 cm by 83% and 183% for furrow and sprinkler-irrigated soils, respectively. At 30–60 cm and 60–90 cm depths, furrow irrigation increased Cd concentration by between 4 and 12% while sprinkler-irrigated soils had significantly ($P < 0.05$) lower (11 and 17%) Cd than the control. The results were unexpected, considering that trace metals undergo adsorption. This anomaly has been reported by previous researchers on soils amended with a mixture of sludge and effluent (e.g. Oloya and Tagwira 1996a). This could be attributed to plant uptake and leaching under acidic soil conditions. Mobility could

also have been enhanced by high hydraulic loading of approximately 4500 mm ha⁻¹ yr⁻¹. A study conducted by Henning et al. (2001) on a sludge-amended sandy soil showed higher Cd concentrations in control than sludge-amended soils due to Cd mobility. Kabata-Pendias and Andriano (1995) also reported trace metal mobility on acid sandy soils of low organic matter. Moreover, plants can take up and accumulate Cd without showing phytotoxicity symptoms (Pennsylvania State University 1985).

Zinc concentrations showed the same trend as Cd with average concentrations (mg kg⁻¹) of 118–120, 168–183 and 135–138 for the control, furrow and sprinkler-irrigated sites, respectively, but concentrations were within acceptable ranges of 150–300 mg kg⁻¹. However, these concentrations were relatively higher than the ranges (0.57–36.9 mg kg⁻¹) reported by Rattan et al. (2005) and Nyamangara and Mwezewa (1999). This disagreement was due to different extraction methods—the double acid extraction used in this study was stronger than the extract used in their studies. Studies that used the double acid method obtained results within comparable ranges (e.g. Reemtsma et al. 2000; Mapanda et al. 2005). However, Zn is most readily absorbed by plants than any other trace metal and has a high potential to move beyond the depth of incorporation (Kabata-Pendias and Andriano 1995). This was further supported by Oloya and Tagwira (1996a) who reported low zinc in effluent-irrigated clay soils which did not correspond to the loading rates of Zn. They attributed this to the leaching of Zn by large volumes of irrigation water.

Copper concentrations (mg kg⁻¹) for the control, furrow and sprinkler-irrigated sites ranged from 7.3 to 22.8, 15.1 to 25.8 and 17.5 to 27.2, respectively. Both irrigated sites had significantly ($P < 0.05$) lower concentration than the control at 0–30 cm depth probably due to plant uptake and mobility. Significantly ($P < 0.05$) higher Cu concentrations at 30–60 and 60–90 cm depths further confirmed the mobility of Cu under acidic conditions. Copper concentrations in the ranges observed have been reported on wastewater-irrigated soils by Mapanda et al. (2005). All Cu concentrations were below the maximum limit of 300 mg kg⁻¹.

Contrary to Cd, Zn and Cu, effluent irrigation significantly ($P < 0.05$) increased soil Cr concentration compared to the control. Average Cr concentrations (mg kg⁻¹) increased in the order;

furrow-irrigated (16.8–18.3) > sprinkler (13.5–13.8) > control (11.8–12.0). On average, effluent-irrigated soils had between 8 and 55% higher Cr concentrations than control. This conflicted with Mapanda et al. (2005) who observed higher Cr concentrations in control than wastewater-irrigated soils. They offered no convincing explanation for the anomaly. There was no significant change in Cr concentration with depth, although a slight increase was observed at 30–60 cm depth for all treatments. Cr is not an essential plant nutrient hence no substantial uptake was expected, leading to the accumulation in effluent-irrigated soils. No guideline limits for wastewater-irrigated soils have been derived for Cr. However, the observed concentrations were within typical ranges of between 5 and 1500 mg kg⁻¹ by Landon (1991) for uncontaminated soils.

Conclusions

Chemical analysis of effluent showed that most parameters were within acceptable limits for wastewater irrigation. Our results revealed that apart from being a source of irrigation water, effluent contained adequate plant nutrients, resulting in significant ($P < 0.05$) soil enrichment with essential nutrients for plant growth, which are commonly deficient in most soils of Zimbabwe. Therefore, effluent irrigation offers an opportunity to increase and stabilise crop yields in smallholder farming systems where low rainfall and declining soil fertility contribute to low crop yields. This is particularly so for poor households depending on peri-urban crop production to diversify food security and livelihoods options. However, the build-up of Cr and the depletions and mobility of Zn, Cd and Cu could threaten the sustainability of effluent irrigation. We recommend a review of irrigation practices to ensure that effluent loading rates are based on crop water requirements, effluent quality and environmental considerations.

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