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Treating and reusing polluted runoff from an informal settlement, South Africa

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ABSTRACT

Biofiltration holds one of the most promising options for removing environmental pollutants from water by reducing inorganic matter, and nutrient concentrations and removing pathogens. This study evaluates the performance of six large field-scale biofiltration cells and assesses the risk of reusing this treated water for irrigating food gardens. The study took place at an abandoned wastewater treatment work (WWTW) in Franschhoek, South Africa. A batch operation was used to measure physical water properties and nutrient concentrations. Large stone cells performed best in reducing ammonia nitrogen (NH_3) and orthophosphate (PO_4^{3-}) by 98 and 95%, respectively, however, an overall increase in nitrate (NO_3^-) and nitrites (NO_2^-) was also observed in these and other cells. Phytoremediation made a marginal contribution to reducing contamination. The extent to which biofiltration can be used to clean and reuse contaminated surface water runoff from an informal settlement to safely reuse the water for irrigation purposes is poorly understood. Laboratory analyses revealed that the water quality from four successive harvests broadly met South African guidelines for irrigation and compared favourably with the quality of vegetables from local supermarkets.

Key words: informal settlements, nature-based treatment, nutrient loading, risk assessment

HIGHLIGHTS

- Biofiltration treatment of contaminated runoff from informal settlements holds considerable promise in reducing nutrient concentrations.
- Large stone filter media performed best in reducing nutrient concentrations by 95%.
- Treated water for vegetable irrigation complied with South African water quality standards.
- The quality of vegetables compared favourably with similar vegetables sold in local supermarkets.

INTRODUCTION

Informal settlements in South Africa are characterised by densely populated residential areas that are found on land that is often unsuitable for residential areas as they are situated in low-lying, flood-prone areas with limited access to basic public services such as water, sanitation and drainage (Armitage *et al.* 2009). After the country's first democratic election in 1994, the newly elected government began by repealing unjust, spatial planning laws that had been used to uphold the Apartheid legislation. Since then, rapid urbanisation has led to the growth of informal settlements in most major cities and towns. For example, the Group Areas Act of 1950 prohibited black South Africans from owning private land. Once this was repealed, people were able to move from rural and peri-urban townships to formal urban areas in search of better access to public services, employment and housing opportunities. The national government attempted to accelerate the construction of social housing but found it difficult to keep up with the rising demand (Turok & Borel-Saladin 2014). By 2018, nearly 1.4 million households or approximately 3.6 million people were still living in informal settlements (SERI 2018). Residents of informal settlements have limited access to sanitation, sewerage and greywater disposal services, which leaves them with little choice but to dispose of their waste into makeshift drainage channels close to their dwellings. Greywater accounts for most of the discarded household water, which flows as surface runoff that can enter downstream water bodies and infiltrate soil, resulting in groundwater contamination. According to the best available estimates, over 490,000 m^3 of greywater is generated every day in South Africa's informal settlements (Carden *et al.* 2007). In addition, greywater runoff frequently combines with blackwater that flows from dysfunctional or leaking toilets and open defaecation. Studies have found that chemical oxygen demand (COD) concentrations in runoff from informal settlements ranged from 1,500 to 8,500 mg/l; oil and grease concentrations ranged from 30 to 2,000 mg/l; electrical conductivity (EC) from 50 to 1,500 mS/m; and that bacteriological colony-forming units (CFU/100 ml) were similar to untreated sewage (Armitage *et al.* 2009). The responsibility for delivering services to informal settlements is left largely to local governments that are required to provide and maintain basic public services, often without adequate resources.

This paper presents the results of a field-scale study that demonstrates the application of biofiltration as a bioremediation process to clean contaminated surface water runoff from an informal settlement. The paper focuses attention on the use of nature-based solutions (NbS) in reducing nutrient concentrations. The 2-year study evaluates the performance of six large biofiltration cells that are packed with natural media to treat contaminated water and assesses the risk of reusing the water to irrigate food gardens. This study examines the extent to which nature-based bioremediation processes are capable of reducing the nutrient load through the abatement of particles in the biofilm, and nitrification and denitrification processes so that water can be used safely to improve food security at a decentralised site.

NATURE-BASED TREATMENT OF WATER

Arguments for advancing nature-based treatment processes in developing countries generally recognise the need for low-cost infrastructure that is passive and easy to construct, and demands limited skills to operate and manage (Zhang *et al.* 2014). South Africa provides an ideal opportunity to showcase the benefits of nature-based treatment. The country faces many challenges regarding access to public services, over-exploitation and pollution of water resources, and leakage and wastage in some urban water distribution systems, especially in catchments occupied by informal settlements (Armitage *et al.* 2009; Carden *et al.* 2018).

Biofiltration holds one of the most promising options for removing environmental pollutants from water (Oral *et al.* 2020) by reducing inorganic matter and nutrient concentrations and removing pathogens (Saeed & Sun 2012). This study examines the design, operation and performance of bioremediation in the form of a nature-based filtration system for treating contaminated water without the addition of chemicals. Biofiltration systems are designed to reduce sedimentation, enable chemical precipitation and adsorption by plants and provide a rich habitat for microbial organisms (Kadlec & Wallace 2008). The extent to which filtration becomes increasingly efficient or optimal in the removal or reduction of pollutants depends on several factors including the packing media used in the filters, the quality of the feed or influent water, variability of flow rate, retention time, the extent of the biofilm, the simultaneous function of nitrification and denitrification (SND) and local environmental conditions (Samsó & Garcia 2013; Oral *et al.* 2020). Some researchers claim that the type of filtration media is most important for controlling the physical (filtration, sedimentation), chemical (adsorption) and biological removal (microbial decay) of pollutants (Davis *et al.* 2006; Hatt *et al.* 2007). The growth of bacterial communities within the filtration media plays a particularly important role in the transformation and degradation of pollutants; however, this depends on the availability of substrate (i.e. carbon), which is influenced by various factors including the hydraulic loading rate (HLR) and influent feed (Samsó & Garcia 2013; Wu *et al.* 2022). Others claim that vegetation is more important because it promotes pollutant removal through phytoremediation and evapotranspiration (Chandrasena *et al.* 2014; Rycewicz-Borecki *et al.* 2017). However, it was also noted that the design ultimately depends on the target pollutant in question (Ilyas *et al.* 2020).

This study examines the performance of biofilters to reduce, degrade and adsorb phosphorus (P) and nitrogen (N). Three paired cells were planted with indigenous reeds while the other three were left without vegetation. The degradation of N varies according to N chemical species with major transformation and removal mechanisms involving ammonification, nitrification, denitrification, plant and microbial uptake, adsorption, desorption, NH_3 oxidation and leaching (Hauer & Lamberti 2017). Denitrification is usually the dominant N removal process in mature biofilters; however, alternative microbial N removal processes include NH_3 oxidation in biofilters in the treatment of NH_4^+-rich wastewater (Tanner 2004). In general, sub-surface biofilters have a high potential for NO_3^- reduction due to the presence of anaerobic conditions, heterotrophs and nitrifiers, especially when this type of filtration configuration is combined with several removal mechanisms including sedimentation, fine filtration, adsorption and biological uptake (Kadlec & Wallace 2008; Wu *et al.* 2022).

Research findings indicate that infiltration is a key factor in controlling both hydrologic and treatment performance (Brander *et al.* 2009) in which the capacity of biofilters is a function of the void space that is influenced by the media texture, structure and the roots of plants growing in the medium (Skorobogatov *et al.* 2020). Water movement in the filtration media significantly affects the bacterial community, and hence, is highly influential in both the hydrologic and treatment performance of biofilters. Although clogging is often cited as a major hindrance to biofiltration performance, some authors have argued that it can be managed in such a way that it maintains, or even enhances, performance (Samsó & Garcia 2013; Wu *et al.* 2022). This is explained by the growth of an active biofilm in the clogging development zone, which promotes simultaneous nitrification and denitrification in the biofilter.

The degree of phosphate adsorption by granular media is primarily determined by its texture and grain size distribution, as well as the iron (Fe) and, to a lesser extent, the aluminium (Al) and calcium (Ca) content (García *et al.* 2005). Loose gravel or stone aggregates are widely used as biofilter media because it does not clog easily; however, it only adsorbs small amounts of P because of their coarse texture and low concentrations of Fe and Al. In addition, the binding sites of gravel generally become saturated within several weeks or months after establishment, which reduces the potential effect of adsorption (García *et al.* 2005). Several attempts have, therefore, been made to improve the sorption of P, which is largely particle-bound, by amending the media with various reactive materials, including iron-enriched sand (Erickson *et al.* 2012), fly ash (Kandel *et al.* 2017), water treatment residuals (Lucas & Greenway 2011), lime and alum sludge (Adhikari *et al.* 2016). Although these amendments have improved P removal, N removal is not as straightforward. The removal of N is primarily determined by a combination of plant-soil interactions and microorganisms in N speciation and removal (Skorobogatov *et al.* 2020). In addition, the dominant species of N in biofilters usually include NO_3^- and dissolved N, which are both poorly retained by the media (Liu & Davis 2014).

Hydraulic parameters such as HLR and hydraulic retention time (HRT) are also cited as the important factors in controlling contaminant decay or removal in biofilters (David *et al.* 2022). In general, a higher HRT (and lower HLR) promotes microbial degradation and the sorption of pollutants while lower HRTs result in less removal and may lead to clogging. Biofilter effluent water quality is largely dependent upon the HRT of the wastewater within the biofilm development zone. In general, a higher HRT (and lower HLR) promotes microbial degradation and sorption of pollutants to the filtration media, whereas shorter HRTs (and higher HLRs) can lead to effluent quality impairment by forcing removal to occur in less efficient layers below the biofilm development zone. As a result, conservative HLRs can improve effluent quality by concentrating the removal burden closer to the more efficient biofilm development zone.

RESEARCH SITE

The study took place at an abandoned wastewater treatment work (WWTW) in Franschhoek, South Africa, on a site that was dubbed the 'Water Hub' by researchers in 2018. The original Franschhoek WWTW was constructed in the 1960s on land that was purchased by the local authority. The capacity of the treatment works exceeded its designed in the early 2000s, and despite efforts to upgrade the plant in 2010, a decision was taken to divert the sewerage system to a new WWTW situated approximately 5 km north of the current location. Thus, the abandoned property provided an opportunity to build an innovative research project and build the concept of a resource recovery centre as opposed to treating waste. The project began with the treatment of contaminated surface water that was largely caused by runoff from an informal settlement and was being discharged into the Stiebeuel River resulting in a highly polluted river. In 2016, the Western Cape provincial government, which is a tier of national government, advertised a public tender for the development of the site. The terms of reference focused mainly on the development of a business plan for the site but included an allowance for rudimentary interventions and modifying the existing infrastructure for the treatment of water for research purposes. The contract was granted to Isidima, a small consulting engineering firm. As part of the project's initial phase, the consultants, in collaboration with the University of Cape Town's Future Water Institute, designed and retrofitted six sludge drying beds and converted these into biofiltration cells. Once the tender project was completed in 2018, the Future Water Institute took responsibility for continuing to manage the site and for initiating new research projects at the Water Hub.

The location of the site and surrounding conditions are representatives of many peri-urban places in South Africa in which the socio-economic context of informal settlements co-exists near wealthier property owners. The site is situated in the vineyards of the Franschhoek valley which is also a tourist destination for thousands of local and international visitors each year who come to enjoy the scenery, and the opportunity to taste local wines and high-quality cuisine that is served on wine estates and in the local restaurants. By contrast, the informal residential settlement, situated less than 1 km upslope of the Water Hub, continues to reflect the Apartheid legacy of racial separation, inequality and urban poverty. Around 7,500 people live in this residential area, of which 64% are living in an informal settlement in over 1,800 houses and share 150 waterborne communal ablution toilets (Stellenbosch Municipality 2021). The formal town of Franschhoek is approximately 2.5 km from the poorer urban areas of Langrug and Groendal (Figure 1).

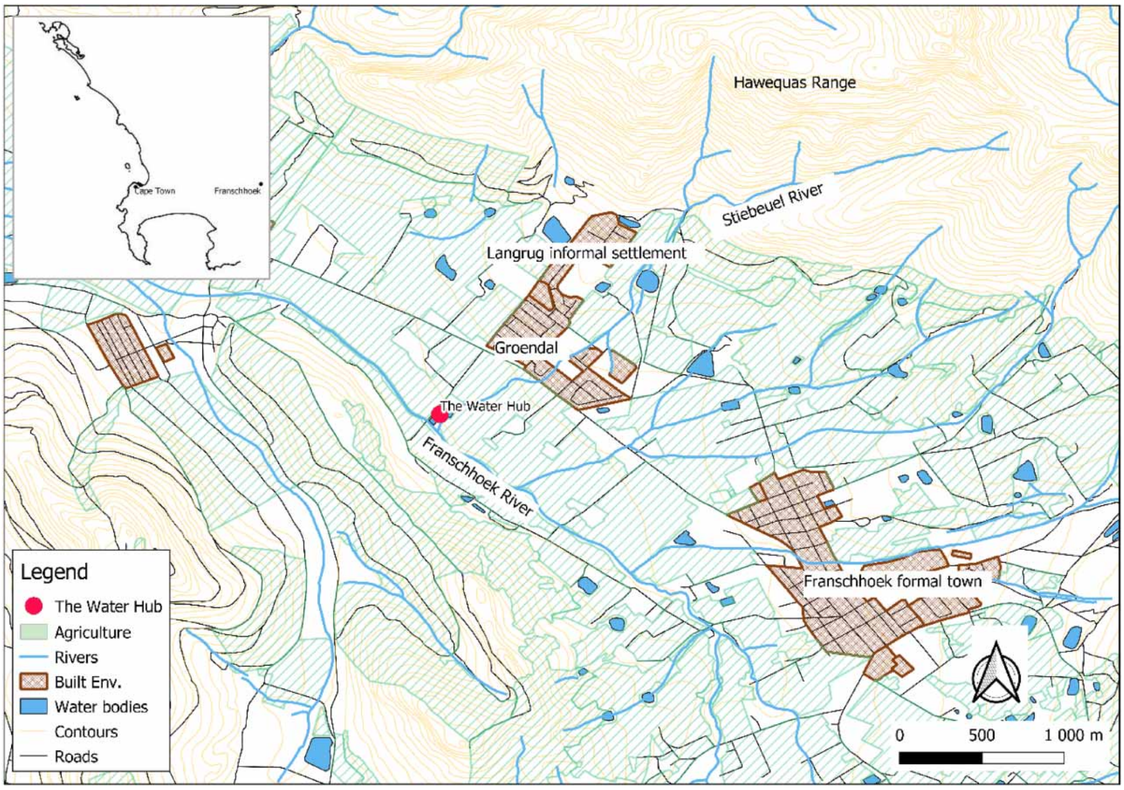

Figure 1 | The location of Franschhoek and the Water Hub in relation to Langrug informal settlement and the formal town (Data source: National Geo-spatial Information, DALRRD, South Africa).

The Franschhoek valley has a Mediterranean climate with seasonal winter frontal rainfall and warm, dry summers. About 80% of the rainfall falls between April and September with an average annual rainfall of 784 mm in the lower valley and 903 mm in the upper parts of the valley in the watershed of the Berg River catchment (Fell 2018). Temperatures can reach 45 °C during the dry summer months of January and February, with the highest temperatures occurring between October and March (Fell 2018). The Stiebeuel River drains a small catchment area of 4.69 km^2 catchment area. It is a perennial stream that is fed by springs, rainfall, surface water from the two main settlements, Groendal and Langrug and discarded greywater and dysfunctional sanitation systems. The river rises in the Hawequas Mountains and flows at a 1:12 gradient through loamy sand and clayey soils (Armitage *et al.* 2009).

RESEARCH DESIGN

The first phase of the project began with the conversion of six drying beds which were repurposed into biofiltration cells, each measuring 16 m × 3 m × 0.63 m (Figure 2). Each cell was lined with a 4 mm low-density

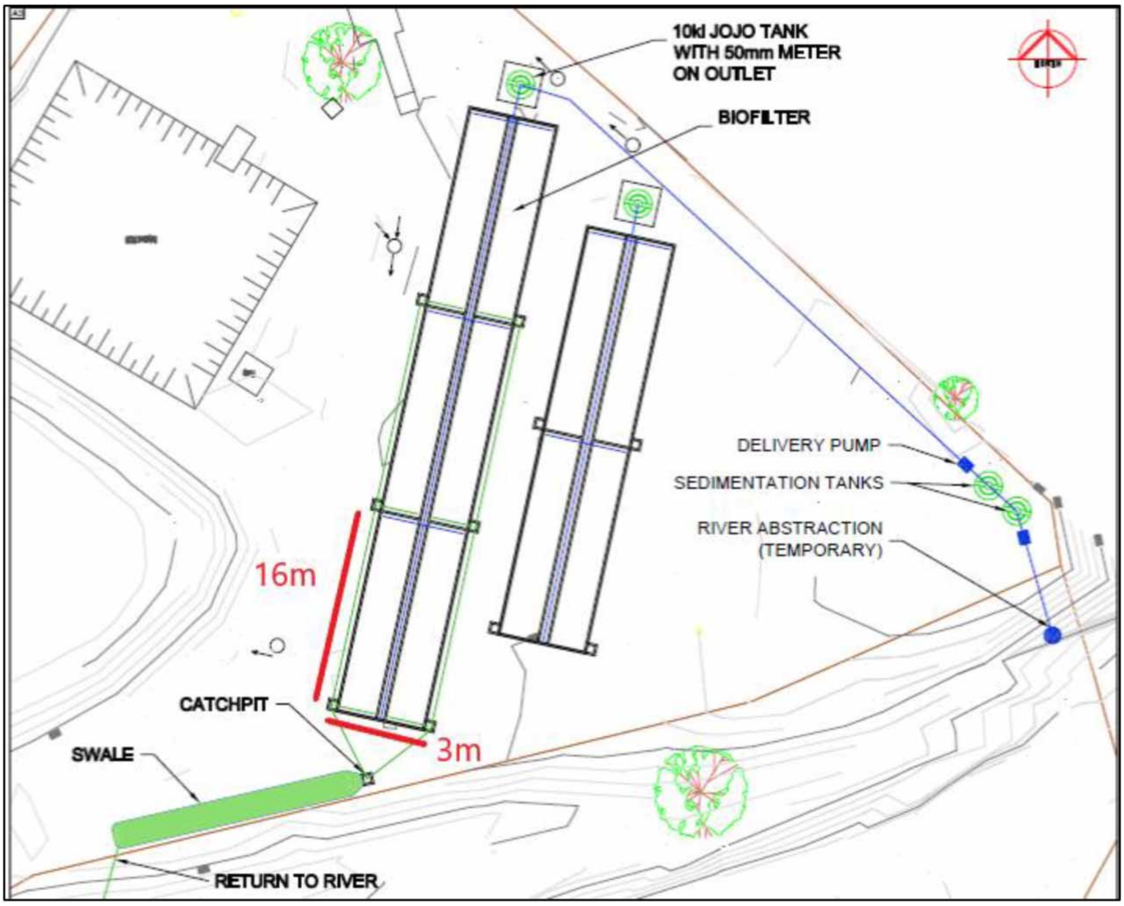

Figure 2 | Design of biofiltration cells developed by Isidima Design and Development Company in 2017.

polyethene material and filled with media consisting of large stone aggregates (19–23 mm), smaller stones (7–9 mm) and peach stones (pips), the latter chosen as a carbon source and for its uneven surface area (Figure 2). One cell from each pair was planted with a mixture of *Typha capensis* and *Phragmites australis* reeds to compare the effectiveness of phytoremediation against those without vegetation.

Two 10 kl water tanks were installed at the head of each set of biofilter cells as temporary storage of raw water that was abstracted from the Stiebeuel River (Figure 2). The biofiltration cells were operated as a batch-fed system which involved regular fill and draw cycles similar to that of a sequencing batch reactor. The cells were filled on day 1 in the 'fill' phase ($\pm$2 h) and remained saturated in the 'react' phase (7 days) until they were emptied in the 'draw' phase ($\pm$2 h). This operation mode was selected because it requires a lower power input than a continuously fed system and is thus more appropriate in the context of informal settlements where electricity supply is often limited. A batch-fed system also maximises the amount of time that the water spends in the biofiltration cells which enhances its interaction with the filter media, vegetation and microorganisms. At times, the available energy for pumping water was limited because weather conditions reduced the capture and storage of solar energy. During cool, cloudy conditions, it would often take 3–5 days to fill the two 10 kl tanks using a 1 kWh water pump. Once the water was released into each cell and filled, the capacity of each cell could contain between 11 and 18 kl of water depending on the type of packing material.

Each cell was fed via a network of manually operated valves and reticulated through a series of 40 mm plastic pipelines. A perforated horizontal pipe (inlet) and the valve were used to control the flow and distribution of the water across the breadth of each cell. The discharge point was fitted with a 110 mm U-bend pipe which was used to maintain the height of the water in each cell to within 10 cm of the surface of the packing material. Water samples were collected each week from the respective cells, the inlet tank and the river. The sampling procedure involved removing the arm of the U-bend and releasing about 200 l of water before collection, followed by a full discharge back to the river via a swale channel. A small quantity of water was diverted to a 400-l storage tank and used for irrigating experimental food gardens.

RESEARCH METHOD

Water samples were taken every week or at least bi-monthly during the study period from 2017 to 2019. The average HRT for each cell was between 5 and 7 days. After each HRT, the cells were drained and recharged at an extremely low HLR of approximately 1.5 kl/h. Flow rate is dependent on a gravity feed and hydraulic head of the influent tank. The paper details the first phase of the study, which examined the performance of each cell and compared the results to the feedstock of raw water that was abstracted from the river and briefly held in the storage tank before being discharged to the respective cells. The investigation from August 2017 to November 2019, spanned three consecutive winter and summer seasons. During the study pH, total dissolved solids (TDS), EC, oxidation-reduction potential (ORP), dissolved oxygen (DO), ammonia nitrogen (NH_3), orthophosphate (PO_4^{3-}), nitrate (NO_3^-) and nitrite (NO_2^-) were sampled and analysed. Nutrient samples were analysed using a HACH DR 2700 spectrophotometer and HACH reagents using standard analytical methods as described in the HACH Water Analysis Handbook (5th Edition). The samples were collected in clean 250 ml containers that had been washed, rinsed and often sterilised. Physical water parameters of EC, pH and DO were measured in the field using calibrated handheld sensors.

RESULTS

There are significant challenges in treating water quality that is generated from the informal settlement and densely populated housing area upstream of the Water Hub. The water quality varies with changes in environmental conditions and human activities. Given the time lag to abstract and distribute water from the river to the feeder tanks, the water quality in the storage tanks represents a composite sample. The overall aim was to compare the relative performance of individual biofiltration cells to comply with general standards for irrigating vegetables that in turn would meet norms and standards for human consumption of the produce. The latter was determined by comparing guidelines and recommendations of the World Health Organization (WHO) and the South African Water Quality Guidelines (DWAF 1996) for the reuse of water for irrigation (Rodda *et al.* 2011).

The box and whisker plots (Figure 3(a)–3(d)) illustrate the relative performance of each cell in the reduction of nutrients during the study period followed by an explanation in each case. The figure shows abbreviations for the

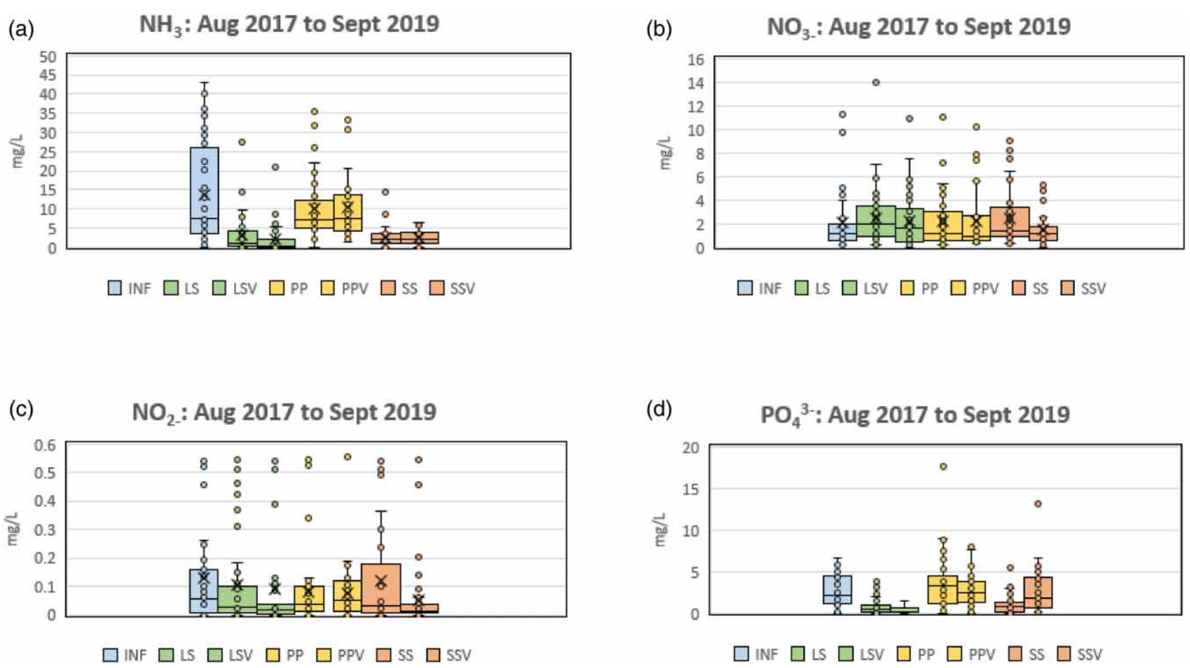

Figure 3 | (a) Ammonia removal efficiency in each cell, (b) nitrate removal efficiency in each cell, (c) nitrite removal efficiency in each cell, (d) phosphate removal efficiency in each cell.

water supply and respective cells as follows: Influent (INF); Large Stones (LV), Large Stone Vegetated (LSV), Peach Pips (PP), Peach Pips Vegetated (PPV), Small Stones (SS) and Small Stone Vegetated (SSV).

Ammonia

The LSV cell was the best-performing cell with the lowest median concentration of NH_3-N at 2.74 mg/l followed by SSV of 2.88 mg/l (Figure 3(a)). On average, LSV showed a 78% reduction of NH_3-N and a 100% reduction rate on three occasions: 19 January; 30 March and 23 April 2018. The PP and PPV cells had the lowest percentage of NH_3-N removal compared to the other cells. An average of 16 mg/l was measured in the influent supply for the whole sampling period (11 August 2017–06 June 2018) with elevated concentrations at the 90th and 95th percentiles of 37.64 and 40.77 mg/l, respectively. Nitrification by microorganisms on the larger surface area of media is a plausible reason for the conversion of ammonium to nitrate (NH_4^+-N to NO_3-N). This finding is consistent with a similar study by Liu *et al.* (2020). In addition, Kawan *et al.* (2022) demonstrated that a gradual decrease in NH_3-N concentration (as shown in the LSV, LS, SSV and SS in this study) is correlated with an increase in NO_3-N concentration.

Nitrate

The assimilative capacity of vegetation combined with tight pore spaces in the SSV cell resulted in a slightly reduced ammonia concentration with a median value of 1.84 mg/l at the point of discharge compared to the influent concentration of 2.22 mg/l (Figure 3(b)). However, there was an increase in the NO_3-N levels in all other cells and observed in the median values of PPV (2.86 mg/l), PP (2.87 mg/l), LSV (2.87 mg/l), LS (2.59 mg/l) and SS (2.87 mg/l). The increase in the nitrate concentration is attributed to the oxidation of NH_3-N to NO_3-N in aerobic conditions (Egea-Corbacho *et al.* 2019). Previous studies have also highlighted the role of organic matter in NO_3^- production (Davis *et al.* 2006; Bratieres *et al.* 2008). Thus, the high NO_3-N outputs could also be associated with the breakdown of organic matter (due to the poor uptake of organic matter by the microbial community). Nitrate is a highly mobile anion that can be removed via anaerobic denitrification to achieve N removal from the system. Nitrate is removed by a single-stage microbiological process under oxygen-poor conditions via the following reaction:

$$4NO_3^- + 5C + 2H_2O \rightarrow 2N_2 + 4HCO_3^- + CO_2$$

Nitrite

The SSV cell had the highest nitrite removal with a median value of 0.06 mg/l followed by PPV (0.09 mg/l), PP (0.1 mg/l), LSV (0.12 mg/l), LS (0.12 mg/l) and SS (0.12 mg/l) (Figure 3(c)). Vegetation adsorption in the SSV and PPV cells resulted in a relatively minor increase in the removal of NO_2^-. Relatively higher maximum concentrations were observed in all cells which could be attributed to the mobility of NO_2^- and leaching (Dlamini *et al.* 2021) as well as the nitrification under aerobic condition. The stability of the NO_2^- and NO_3^- production suggests that nitrification was established during the study period. The relationship between wet weather conditions and an increase in NO_2^- concentration was not established in this study.

Relatively higher NO_2^- removal efficiencies were recorded after three sampling events over 2 months between 1 September 2017 and 27 October 2017 with an average removal efficacy of over 90%. The nitrite removal efficiency was 100% on consecutive sampling events from 8 September until 22 September 2017 due to a higher denitrification rate. Overall, there was a relatively poor NO_2^- removal between January and February 2018 with average removal efficacy of about 16.8% possibly due to the loss of microbial capacity which decreased the rate of denitrification.

Orthophosphate

The highest orthophosphate removal rates were recorded in LSV, SS and LS with the following median values 0.71, 0.83 and 0.89 mg/l, respectively (Figure 3(d)). These best-performing cells showed a removal or abatement of PO_4^{3-} of 75.1% (LSV), 66.8% (SS) and 65.7% (LS). LSV showed the lowest 95th percentile concentration of 3.60 mg/l, followed by LS (3.78 mg/l) and SS (4.75 mg/l). Removal of 100% was recorded on the 15 and 22 September 2017 (LSV), 19 January 2018 (SS) and 15 December 2017 (LS). The LSV was the first cell to show the total removal of PO_4^{3-} followed by LS and SS biofilter cells. A combination of the large surface area and vegetation is likely to have provided the substrate for the removal. The 90th and 95th percentile ranged at 5.43

and 5.88 mg/l, respectively. These levels are higher compared to the <2 mg/l recommended by WHO guidelines for the safe use of wastewater and lower <10 mg/l than recommendations for the South African Water Quality Guidelines for agricultural and irrigation water use.

PHYSICAL PARAMETERS

EC effluent

The water quality of the influent was characterised by elevated EC concentrations with a median value of 544.10 μS/cm typical of sewage and soluble salts found in the Stiebeuel River. The highest EC values were observed between January and April 2018 in a period that is characterised by dry, warm conditions and low flow in the river. The EC median values for the influent (897.63 μS/cm) and LS (269.81 μS/cm) during the summer compared to median values of the influent (263.58 μS/cm) and LS (216.50 μS/cm) during cooler conditions from August to November 2017.

A study by Jeong *et al.* (2016) has shown that EC values below 700 μS/cm are suitable for irrigation, whilst EC values above 3,000 μS/cm can compromise crop growth. These researchers claim that the irrigation water quality limits for irrigation of food and processed food crop should not exceed 700 and 200 μS/cm, respectively. In South Africa, guidelines for EC in irrigation water are ideal if ≤300 μS/cm, acceptable between >300 and ≤500 μS/cm, tolerable if between >500 and ≤850 μS/cm and unacceptable if >850 μS/cm (DWAF 1996).

pH

The pH values throughout the sampling period in the feeder tank (INF) were consistently above 7, with an average of 7.39. The minimum and maximum pH values were 6.6 and 7.8, respectively. The highest median pH values were recorded in the influent (7.39), followed by LSV (7.16). An average surface water pH value of 7.39, provided it remains between the range of 6.5 and 8.4, is acceptable according to South African Water Quality Guidelines for Irrigation, Soil and Irrigation Infrastructure. The pH below or above this limit is set to cause a nutrient imbalance, and toxic ions and can cause corrosion of irrigation infrastructure (Jeong *et al.* 2016).

The LSV and LS cells were the only biofilter cells in which the average pH values were within acceptable ranges at 7.15 and 6.92, respectively. The PPV, PP, SSV and SS had the following average pH values of 6.26, 6.17, 6.43 and 6.35 which are slightly unacceptable (DWAF 1996). PP had the lowest pH value of 4.91 followed by PPV with a value of 5, both of which fall outside of the acceptable range. The acidity levels in PPV and PP coincided with the reduced removal of PO_4^{3-} and NH_3 concentrations. In a study by Kasak *et al.* (2018), it was found that the adsorption capacity of phosphorous is influenced by pH, including Fe, Al and Mg, suggesting that a more acidic pH lowers the phosphorus adsorption capacity of the substrate while alkaline conditions are more likely to increase the adsorption capacity of phosphorous. The decomposition of the peach stones in the PPV and PP cells may be responsible for more acidic conditions and a higher rate of nitrification compared to other cells (Gao *et al.* 2017).

Dissolved oxygen

The lowest median DO values were observed in the PP (1.61 mg/l) followed by PPV (1.86 mg/l). The influent feeder tank had the highest median DO value of 3.23 mg/l. Lower DO median values were recorded in the LSV (2.39 mg/l), LS (2.18 mg/l), SSV (2.11 mg/l), SS (2.09 mg/l), PPV (1.86 mg/l) and PP (1.61 mg/l). The lowest DO values in PP and PPV correlate with lower pH values as described in the previous section.

The lowest DO levels with median values of 1.86 and 1.61 mg/l were recorded in the PPV and PP, respectively, which can be attributed to an increase in DO consumption by NH_3 oxidisers and microorganisms. Other major contributors to the reduction in DO observed by Gao *et al.* (2017) are a microbial population that colonised the spherical-shaped media and larger surface areas that contributed to an increase in oxygen consumption. A further observation indicates that the transportation of atmospheric oxygen into the rhizosphere is shown by higher average DO values in vegetated cells LSV (2.39 mg/l) and PPV (1.82 mg/l) compared to non-vegetated cells of LS (2.18 mg/l) and PP (1.60 mg/l).

COMPARISONS OF CELL PERFORMANCE

The non-parametric Wilcoxon signed-rank test was used to compare the overall removal of nutrients against the influent and between vegetated and non-vegetated cells. Table 1 shows a significant difference in the removal of

Table 1 | Wilcoxon signed-rank (non-parametric test) test scores for each biofilter cell

Biofilter cells	NH_3-N	NO_3-N	NO_2-N	PO_4^{3-}	EC	pH	DO
Influent vs LSV	0	217	197	27	194,5	74	97
Influent vs LS	8	217	186,5	21	67,5	23	85
Influent vs PPV	113	276,5	143	224	0	0	71
Influent vs PP	97,5	331,5	114	333	3	0	54,4
Influent vs SSV	9	130	47	218	107,5	0	157
Influent vs SS	16,5	224	93	6	12,5	0	81
Vegetated vs non-vegetated cells							
LSV vs LS	135	332	295	228,5	183,5	62	261,5
PPV vs PP (25 data set)	56	149	135,5	28	202,5	107,5	90,5
SSV vs SS (25 data set)	109	104,5	110	21	36	135	176,5

Notes:	Significance at 0.005 probability level (Critical values of **168** and **60** for 39 and 25 data samples respectively as per the Wilcoxon Signed-Rank Table)
>168/60(0.005)	There is no significant difference between the biofilter cells removal efficacy
< 168/60(0.005)	There is a significant difference between biofilter cells removal efficacy

incoming NH_3 from all the biofilter cells (LSV, LS, PPV, PP, SSV and SS). However, there is no significant removal of NO_3-N between the incoming concentration and observed concentrations in the biofilter cells (LSV, LS, PPV, PP, SSV and SS). The concentration of NO_3^- increased in the biofilter cells with the oxidation of NH_3^- (Liu *et al.* 2020).

The LSV was the best-performing cell in the removal of NH_3 followed by PO_4^{3-} with 98 and 95% median percentages, respectively. LS is the second-best-performing cell in the removal of NH_3 followed by PO_4^{3-} with the following median percentages, 75 and 80%, respectively. A significant difference in the NH_3 removal was observed between LSV and LS, whilst there was no significant difference between LSV and LS in the removal of PO_4^{3-} (Table 1). The PPV and PP cells had the lowest removal percentages of NH_3 and PO_4^{3-} compared to other cells. A possible explanation was found in a study by Loh *et al.* (2021) in which lower nitrification efficiency and higher denitrification rates occurred in biofilters characterised by organic carbon sources and low oxygen concentration that encouraged the growth of heterotrophic denitrifies. A median percentage of >60 in the removal of NH_3 was achieved in both the SSV and SS cells and over 80% was achieved in the removal of NO_2^- and PO_4^3 in the SSV and SS, respectively. There was a significant difference in the removal of PO_4^3 between SSV and SS suggesting that the smaller-sized sand properties in the biofilter improved the contact between PO_4^{3-} and the biofilm. Overall, the large stone cells showed the most consistent degradation and removal of NH_3 and PO_4^{3-}.

Direct uptake of nutrients by plants might be minimal, but plants also play an important role in promoting aerobic conditions (oxygenation) in biofilters – e.g. during photosynthesis or direct transport from the atmosphere through their stems and roots to the rhizosphere. This could explain the higher NO_3^- removal efficiencies observed in the vegetated cells. Other factors such as microbiological activity in the biofilm on large and small surface stone aggregates appear to be more significant in the removal of PO_4^{3-} and NH_3. This is contrary to findings from a recent study which suggests that vegetation was one of the major biofilter design features for optimal nutrient removal (Zhang *et al.* 2021). However, in the case of the Water Hub, the combination of substrate selection, the role of microbial organisms and habitat, the rate of nitrification and denitrification processes and hydraulic residence time appear to be more important in the removal and reduction of nutrient concentrations. Other features that were recognised in a study by Loh *et al.* (2021) indicate that proficiency is improved when the filter media have larger surface areas, higher porosity, non-toxic, higher adsorption capacity and phenolic hydroxyls and carboxylic substrates.

TARGET IRRIGATION WATER QUALITY

A main objective of this study was to determine whether the treated runoff can be safely reused for irrigation. In a study on the usage and management of greywater in South African informal settlements, Carden *et al.* (2007) showed how water quality runoff from an informal settlement was highly varied and heavily influenced by activities involved in producing polluted water. Their research also revealed that individuals were generally hesitant to use greywater for irrigating food crops, and they attributed this reluctance to cultural and religious beliefs, as well as general impressions of inferior quality. Public concerns and sentiments are significant barriers to the reuse of untreated greywater for the irrigation of food crops (Mzini & Winter 2015). There is also limited information available about the effect of irrigating small-scale crop production on the uptake of chemical concentrations in edible vegetable crops in the reuse of water that is generated from an informal settlement.

In a South African research study, Rodda *et al.* (2011) examined target water qualities for small-scale irrigation crop production based on two sources of guidelines for the safe reuse of water, namely the South African Water Quality Guidelines for Irrigation (DWAF 1996) and the WHO Guidelines for the Safe Use of Wastewater, Excreta and Greywater (WHO 2006). The greywater constituent is presented as a continuum from target water quality ranges to water quality that is not recommended for irrigation (Table 2). This table is used as a reference to compare the treated water that was used to irrigate small-scale vegetable gardens at the Water Hub (Table 3).

Table 2 | Target water quality ranges for small-scale irrigation in South Africa (adapted from Rodda *et al.* 2011)

Greywater constituent	Target water quality range Suitable for unrestricted use with minimal risk to human health, plants or soil	Maximum water quality range (applicable only to well-drained chemically stable soils) Increasing risk to human health, plants or soil	Water quality is suitable only for short-term use on site-specific basis1 Significant risk to human health, plants or soil; tolerable for short-term use only	Water quality is not recommended for irrigation use Excessive risk to human health, plants or soil
Physical and chemical				
Electrical conductivity (µS/cm)	<400	400–2,000	2,000–5,400	>5,400
pH	6.5–8.4	6–9	6–9	<6 > 9
Boron mg/l	<0.5	0.5–4.0	4.0–6.0	>6.0
Total phosphorous mg/l	<10	10–20	20–60	>60
Total inorganic nitrogen mg/l	<10	10–15	15–50	>50
Microbiological				
E. coli (colony-forming units, CFU·100 ml^{-1})	<1	$1–10^3$ (1–1,000)	$10^3–10^5$ (1,000–100,000) *The range can be extended to 10^7 (10,000,000) if irrigation is sub-surface.*	$>10^7$ (>10,000,000)

In this study, treated water from the biofiltration cells was reticulated to a 400-l storage tank each week for irrigation purposes. Although there are limitations in the range of parameters that were identified by Rodda *et al.* (2011) (Table 2), Table 3 indicates that the treated water in this study met physical, chemical and biological ranges for water that was suitable for unrestricted use with minimal risk to human health, plants and soils except for TIN which showed a potential risk. The irrigation tank water is a composite sample from the various cells. Aside from the weekly and bi-monthly sample analysis of each cell, the irrigation water was analysed at the same time when crops were harvested. Water quality ranges in Table 3 represent four grab samples that were taken over 3 years. There are low concentrations of heavy metals which are to be expected from activities in an informal settlement.

Table 3 | Water quality ranges measured at the point of irrigation at the Water Hub

Water quality ranges Irrigation tank	pH	EC μS/cm	Na mg/l	K mg/l	Ca mg/l	Mg mg/l	Fe mg/l	Cl mg/l	PO_4^{3-} mg/l	TIN[a] mg/l	B mg/l	Cu mg/l	*E. coli* cfu/100 ml
Low	6.5	250	30.4	1.9	12.8	<1	0.05	34	0.01	0.02	<0.08	<0.02	<1
High	7.1	310	38.7	7.8	23.6	3.7	0.18	38.5	4.25*	29.2**	<0.08	<0.02	8

[a]Total inorganic nitrogen ($NH_3 + NO_2 + NO_3$), *Mean = 1.04 **Mean = 5.27.

CROP ANALYSIS

From 2018 to 2021, four crop harvests of leafy and bulbous vegetables were collected and analysed for physical, chemical and microbiological content. The analysis was undertaken at Bemlab, a SANAS-accredited testing laboratory by ISO/IEC 17025:2017. Included in each sample, were matching vegetables that were purchased at local supermarket stores namely, Checkers, Pick 'n Pay, Spar and Woolworths for bulbous plants, while leafy plants were compared against norms that were recommended and typically measured by the laboratory. Instead of using average or median values, which do not account for the possibility of excessive contamination, the findings are shown in a range. All vegetables were irrigated by hand using watering cans by sprinkling water on the plants.

Spinach

The analysis of spinach shows that these crops are largely within the recommended norms, except for Na, Mn, Fe, Cu and Zn. Unwashed samples were sent for analysis which may explain the relatively high Na content that was deposited on the spinach leaves because plants were irrigated with an overhead sprinkler system (Table 4).

Table 4 | Chemical analysis of spinach from using treated water for irrigation compared to laboratory norms

SPINACH		N %	P	K	Ca	Mg	Na mg/kg	Mn	Fe	Cu	Zn	B
Water hub	**Low**	4.19	0.45	2.81	0.49	0.56	28,170	20.4	73	2.4	45.4	24
	High	5.64	0.79	6.95	0.93	1.18	39,771	205	1,180	57	1,508	40
Norms	**Low**	4	0.4	3	0.6	0.35		30	200	7	50	40
	High	6	0.6	4.5	1.2	0.8	4,000	60	250	15	75	80

Lettuce

The analysis (Table 5) indicates that lettuce samples were within the norms for all constituents.

Table 5 | Chemical analysis of lettuce from using treated water for irrigation compared to laboratory norms

Lettuce		N %	P	K	Ca	Mg	Na mg/kg	Mn	Fe	Cu	Zn	B
Water hub	**Low**	3.34	0.57	6.49	0.4	0.2	2,680	30	103	5	47	21
	High	4.39	0.73	7.39	1.17	0.32	3,735	213	112	5	52	33.4
Norms	**Low**	3	0.4	5	1.5	0.3	0	60	130	10	25	25
	High	4.5	0.6	8	3	0.6	10,000	300	200	25	70	55

Carrot

On occasions, carrots that were irrigated with treated water were malformed being forked and crooked which is likely to limit consumer appeal and marketability. Malformation of carrot roots is due to compost or manure with excessive nitrogen concentration and exposure to high soil temperatures (Winter 2015). Table 6 shows elevated concentrations compared to carrots sold in local supermarkets in the case of K, Mg, Na, Fe and Zn.

Table 6 | Chemical analysis of carrots from using treated water for irrigation compared to laboratory norms

Carrots		P %	N	K	Ca	Mg	Water% Fruit mass	g/fruit mg/100 g fresh mass	Na	Mn	Fe	Cu	Zn	B
Water hub	**Low**	22.3	54	123	21.6	7.6	87.59	14.9	131	0.62	1.8	0	1.8	1.5
	High	80.76	715	653	69.2	76.6	93.76	137	2,485.8	5.7	45.9	1.7	14.4	3.3
Supermarkets	**Low**	24	61	144	16.8	8	86.59	4.1	146	0.88	1.8	0.21	1.7	1.7
	High	61.24	690	456	78.3	48.8	91.37	119.3	1,481.3	3	6.4	0.7	4.2	2.7

Beetroot

Salt (Na) concentration in greywater can reduce the yield because beetroot is known to be only moderately tolerant of salts (DWAF 1996). In an experimental study by Mzini & Winter (2015), beetroot grown with potable water produced the best quality products, followed by diluted greywater while greywater performed the worst presumably because of an elevated Na concentration. However, in the present study, the results from the analysis suggest that the range in chemical concentrations from treated water and local supermarket suppliers was similar (Table 7).

Table 7 | Chemical analysis of beetroot from using treated water for irrigation compared to laboratory norms

Beetroot		P %	N	K	Ca	Mg	Water% Fruit mass	g/fruit mg/100 g fresh mass	Na	Mn	Fe	Cu	Zn	B
Water hub	**Low**	36.8	270	317	15	25.5	84.34	13.1	289	1.4	8.5	0.73	5.1	2.1
	High	70.6	433	522	32.9	37.5	91.69	59.95	770	2.7	15.2	2.1	13.9	2.7
Supermarkets	**Low**	55.5	205	219	12.1	25.9	79.72	49.1	493	6.8	7.2	0.51	5.2	1.3
	High	80.81	256	412	19.6	54.5	85.92	135	663	12.5	11.3	1	9	3.1

Treated greywater runoff from an informal settlement has the potential to increase yields due to the higher concentration of minerals, as indicated by EC in this study, yet high salt volumes could have negative consequences for yield, as indicated by the elevated concentration of Na. This concurs with similar findings by Mzini & Winter (2015). In the current study, crop analysis is confined to a selection of parameters as presented in the tables. Further studies involve the identification and measurement of the persistence and mobility of heavy metals and compounds found in contaminants of emerging concern (CEC) in the treated water, soil and plants.

CONCLUSION

The results showed that vegetated (LS) and non-vegetated large stone (LSV) cells effectively reduced NH_3 and PO_4^{3-} concentrations in the contaminated water; however, an overall increase in outflow nitrate (NO_3^-) and nitrites (NO_2^-) was also observed in these and other cells. Limited denitrification was detected in the peach pip (stone) cells resulting from a combination of factors including biofilm development, availability of heterotrophs and bacteriological species. The low variation between the cells suggests that plant absorption was only marginally responsible for eliminating contaminants. Except for the peach pip cells, all other cells effectively reduced N and EC concentrations through a combination of SND processes to eventually meet the minimum water quality requirements for irrigation purposes. All processes in the cells created an optimal pH range for irrigation purposes.

The manipulation of treatment processes for the productive use of scarce water resources, without the addition of chemicals, was investigated in this study by using vegetables as a biomarker to determine the effect of using nature-based processes to produce crops that could be safely consumed that was based on a reasonable range of analytical determinants. Laboratory norms and comparisons with local supermarket products indicate that the quality of successive harvests largely met expectations. Further research will focus on the manipulation of soils and treatment processes to reduce excess Mn, Mg, Zn, Cu and Na using an extended treatment train and oxidation processes.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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