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Nutrient balances with wastewater irrigation and biochar application in urban agriculture of Northern Ghana

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Abstract Urban agriculture in developing countries contributes to food diversity and security of the urban population. Its importance will increase in the future because of fast-growing urbanization. Little is known about nutrient fluxes and balances of these high input agricultural systems, which are characterized by high fertilizer use, often combined with wastewater irrigation. Adding biochar to soil has shown the potential to decrease nutrient leaching, increase yields and nutrient use efficiency. Therefore, we installed lysimeters in a multi-factorial field experimental in Tamale, Northern Ghana. The treatments included a control (no amendments applied), biochar at 20 t ha^{-1}, mineral fertilization according to the farmers' practice and a combination of biochar amendment and fertilization. All treatments were irrigated with tap water or wastewater. The results show higher water losses under wastewater irrigation (+ 33%). The addition of biochar had no effects on nutrient leaching, balances or water flux. Leaching losses of nitrogen were around 200 kg N ha^{-1} when irrigation exceeded the crop demands. When irrigation was more appropriate, the leaching rates were 50–100 kg N ha^{-1}. The leaching of Mg and Ca almost doubled in some seasons and negative mass balances under mineral fertilization entailed soil acidification. Nitrogen balances varied strongly depending on the season, irrigation water qualities or fertilization (− 50 to 222 kg NO$_3$-N ha^{-1}). We conclude that the high nutrient load associated with the commonly-practiced wastewater irrigation entails large leaching losses. These cannot be curbed by biochar application and should be accounted for in fertilizer management in urban vegetable production.

Keywords Nutrient leaching · Vegetable production · Smallholder farmer · Soil acidification

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Introduction

Globally, 5.9% of all crop land is located in urban areas (Thebo et al. 2014) and urban agriculture often substantially contributes to the food supply of urban populations, as shown for Kumasi, Ghana where urban agriculture produces 60–90% of all perishable vegetables consumed in the city (Obuobie et al. 2006). Due to the relatively high revenues that can be achieved from these crops, many farmers practice a year-round production unlike traditional farming with fallow periods. Furthermore, poorly adapted fertilizer inputs and irrigation with waste water from open channels are a common practice in many developing countries. For example, Lydecker and Drechsel (2010) estimated that from the 80 Mio l of waste water generated daily in Accra, Ghana, 11.3 Mio l are used for vegetable production.

As shown with national nutrient balances by Stoorvogel and Smaling (1990), crop production in rural Africa is often associated with nutrient mining resulting in negative soil nutrient balances. Cobo et al. (2010) reviewed 57 studies on the subject and found that more than 75% of the field based studies reported negative nutrient balances (less nutrient input than output) for nitrogen (N) and potassium (K) and 57% for phosphorus (P). In contrast, the few studies that examined nutrient dynamics in African urban agriculture show a very different picture. Sangare et al. (2012) found that the annual nutrient supply in urban agriculture of Bobo Dioulasso, Burkina Faso exceeded crop demands by up to 2012 kg ha^{-1} for N, 450 kg ha^{-1} for P and 393 kg ha^{-1} for K. Similar results were reported by Lompo et al. (2012), who found surpluses in annual partial balances [partial balance (kg ha^{-1}) = nutrient inputs (fertilizer and waste water) (kg ha^{-1}) − crop uptake (kg ha^{-1})] in two urban gardens of 2056 and 1752 kg N ha^{-1}, 615 and 446 kg P ha^{-1} and 1864 and 1643 kg K ha^{-1}. Besides, Graefe et al. (2008) reported positive partial annual balances of up to 7312 kg N ha^{-1}, 530 kg P ha^{-1} and 6827 kg K ha^{-1} in urban gardens of Niamey, Niger. In the same study, these authors also reported figures for rural millet fields being as low as − 39 kg N ha^{-1}, − 9 kg P ha^{-1} and − 33 kg K ha^{-1} a year. This underlines the differences between low input rural farming and very high input urban farming systems.

Such high surplus nutrient inputs are questionable in terms of fertilizer costs and unproductive gaseous or leaching losses of plant nutrients. The latter have rarely been examined in urban agriculture, so far. In one of the few studies, Predotova et al. (2011) estimated nutrient leaching in urban gardens of Niamey, Niger with ion exchange resins and found a negligible loss of up to 7.3 kg N ha^{-1} and 0.7 kg P ha^{-1}. However, these data were collected in a year with unusually low rainfall.

Since high nutrient losses are often related to the commonly low effective cation exchange capacities of Sub-Saharan African soils (Bationo et al. 1998), losses could be reduced by adding organic amendments, such as manures or composts (Gonçalves et al. 2007). However, this may only be a short-term effect due to the high mineralization rates of organic materials under tropical conditions. More recently, the use of biochar has been advocated as a recalcitrant soil amendment to increase sorption capacities and thus reduce the leaching of nutrients from the soil (Biederman and Harpole 2013; Steiner et al. 2008). Biochar can be produced from wood and organic wastes through heating in the absence of oxygen. Several efforts have been made to reduce nutrient leaching form soil with the addition of biochar. In a laboratory study, for example, Laird et al. (2010) found a reduction of N, P and Mg leaching with biochar application to a soil fertilized with pig manure. Pereira et al. (2017) conducted a pot experiment in a greenhouse with biochar derived from pine chip and walnut shell. They found a decrease of N leaching losses from the pots for pine chips biochar but not for walnut shell biochar. In another greenhouse experiment, Borchard et al. (2012) observed an enhanced nitrate leaching by 41% in a sandy soil and a decreased nitrate leaching by 17% in a loamy soil with the same biochar made from beech wood. These findings showed that the impact of biochar on nutrient leaching depends on the kind of biochar and the soil type.

For our study, we hypothesize a strong leaching of easily soluble nutrients from wastewater and fertilizer application, especially during the rainy seasons when high water fluxes occur. Biochar is expected to reduce these leaching losses through sorption and increased water retention and may facilitate a more productive resource use. Therefore, the aims of this study were to (1) quantify the contribution of waste water to the total

seasonal nutrient inputs, (2) measure the extent of nutrient leaching in urban agriculture in a field study, (3) determine the effect of waste water irrigation and biochar on nutrient loads in leaching water, (4) calculate nutrient balances for major nutrients (N, P, K), as well as other elements (Mg, Ca, Na) and (5) asses the contribution of nutrient leaching to the total output of the system. For this purpose, we monitored water and nutrient fluxes in seepage water with lysimeters installed in an experimental urban agriculture field trial in Tamale, Ghana for two years or four consecutive climatic seasons.

Materials and methods

Field experiment set-up

The field experiment was installed in the outskirts of the city of Tamale, Ghana in the Sudanian Savanna. Tamale has a long-term mean annual rainfall of 1090 mm and a daily mean temperature of 27.9 °C (DWD 2017). The soil type at the location is a Petroplinthic Cambisol with a loamy silt texture (45.7% sand, 47% silt and 5.9% clay) according to WRB (IUSS Working Group WRB 2014) classification (Häring et al. 2017). Initial soil properties at 0–0.2 m are 0.41% total C, 0.04% total N, a pH of 5.1 (in $CaCl_2$), a bulk density of 1.42 g cm^{-3} and an effective cation exchange capacity of 36.1 $mmol_c$-kg^{-1}. Further details about soil parameters and methods of measurement can be found in Häring et al. (2017). The area of the experiment is used for irrigated vegetable farming since decades because of the availability of water from a military base. Although, it is located in an urban environment, the soil is still similar to a typical savannah soil and not disturbed by anthropogenic substrates. The treatments used in our study were: Control (no amendments applied), a treatment fertilized according to normal agricultural practice of the local farmers, a biochar treatment with 20 t ha^{-1} rice husk biochar and a biochar treatment with fertilization according to the practice of local farmers. The amount of fertilizer was chosen after interviewing the local farmers at our field site and was not following public fertilizer recommendations. The quality of the irrigation water was used as a factor besides fertilization and biochar application. The untreated wastewater effluent from a

military base and tap water were used for irrigation. Irrigation rates were also chosen by the local farmer practice.

In order to avoid irrigation errors (wrong water quality), a split plot design was used. The plots with the same irrigation quality were merged to main plots. Subplots with different biochar application and fertilization rates were randomly allocated inside the main plots. Furthermore, four blocks were used as replications.

The crops were grown consecutively on the experimental field and comprised in the first (rainy) season (April 2014–September 2014), maize (*Zea mays* L.) (only 4 weeks), lettuce (*Lactuca sativa* L.) and cabbage (*Brassica oleracea* L.). In the second (dry) season (October 2014–March 2015), amaranth (*Amaranthus cruentus* L.), lettuce and again amaranth were grown. During the third (wet) season (April 2015–September 2015) nalta jute (*Corchorus olitorius* L.) (two times), amaranth and again nalta jute were planted. During the last (dry) season (October 2015–March 2016), crops were roselle (*Hibiscus sabdariffa* L.), lettuce and carrots (*Daucus carota* L.). The types of crops were chosen together with the farmers.

Fertilization was done with NPK 15–15–15 at rates of 200–563 kg ha^{-1} per crop except for two crop cycles of nalta jute in the second wet season when urea was used at 247 and 256 kg ha^{-1}.

Biochar production and application

The biochar used in this experiment was prepared from rice husks collected from waste heaps of a local rice mill in a custom-made kiln at KNUST, Kumasi, Ghana. Highest treatment temperature was about 550 °C. Rice husk was chosen as the feedstock material since it is an abundant waste product without alternative use in Ghana. The rice husk biochar had a total C content of 42.4%, 0.4% total N, a pH of 9.1 (in water) and 63 m^2 g^{-1} BET surface area (using Brunnauer–Emmett–Teller theory to determine surface area from a N_2 adsorption isotherme). Biochar was incorporated at the beginning of the experiment into the soil of the plots and into the pre-installed lysimeter to a depth of 0.2 m with manual ploughing using traditional hoes. The biochar was 4 weeks old at the time of incorporation into soil.

Lysimeters

In order to measure percolation rates and nutrient leaching, passive wick lysimeters (supplementary material Fig. S1) was constructed from PVC pipes with an inner diameter of 0.28 m, adapted from descriptions of similar instruments in other studies (Brown et al. 1989; Holder et al. 1991; Siemens and Kaupenjohann 2004). The undisturbed soil cores were taken to a depth of 0.4 m before the experiment (biochar application) was established. The top PVC pipe was forced into the soil with hammers and subsequently excavated. Afterwards a PVC plate with a glass fiber wick (Thermo E Glasstrickschnur ST1, H.K.O. Oberhausen, Germany) of 0.4 m length and 10 mm in diameter were installed at the bottom of the tube. The glass fiber wick was used to create a hanging water column and draw water from the soil. After preparation of the tube with the soil core, the tube was placed on top of another tube and sealed with epoxy resin. One lysimeter were installed inside the experimental plot by digging an approx. $0.6 \times 0.6 \times 1.2$ m hole at one corner of the plot and placing the lysimeter inside. Subsequently, the soil surface in the lysimeter and the plots were leveled and the soil around the lysimeter was filled back horizon-wise. The leachate was collected in a bucket in the lysimeter and extracted through a PVC tube at the sampling dates. In total, 36 lysimeter were installed in the 2×4 m experimental plots of the field experiment and received the same treatment (irrigation, fertilization and other management e.g. ploughing) as the rest of the plot. Amounts of biochar, fertilizer and irrigation water was separately weight and applied to the lysimeter. The leachate was sampled weekly apart from times when no rain or irrigation took place. The amount of leachate was measured and a sample transported to the lab in an icebox. All samples were acidified after pH and EC measurement and stored in a fridge until further analysis. Rain, tap and waste water were sampled weekly (when available or when used for irrigation) and treated similarly.

Water analysis

In all leachate, irrigation water and rain samples, pH (pH meter basic 20, Crison Instruments S.A., Barcelona, Spain) and electrical conductivity (Conductivity Meter Basic 20, Crison Instruments S.A., Barcelona, Spain)

were recorded immediately after sampling. NO_3-N (Cataldo et al. 1975), NH_4-N (Koroleff 1976) and PO_4-P (Ohno and Zibilske 1991) were measured photometrically. Light absorption at the specific wavelengths was measured with an UV/VIS spectrophotometer (Pharo 300 Spectroquant, Merck GmbH, Darmstadt, Germany). Total amounts of K, Mg Na and Ca were determined in the acidified samples after transport to Germany using an ICP-OES (Ciros CCD, SPECTRO Analytical Instruments GmbH, Kleve, Germany). Details about measurement of nutrient concentrations in crops are reported in Akoto-Danso et al. (2018).

Statistical data analysis and calculations

To calculate leaching losses per lysimeter sampling we used the following equation:

$$
\begin{aligned}
\text{Leaching loss} \,(\text{kg ha}^{-1}) \\
= \text{concentration of element in leachate } (\text{mg l}^{-1}) \\
\times \text{ amount of leachate (l)} \times 0.162
\end{aligned}
$$

The factor 0.162 was used to account for the area of the lysimeter (0.28 m in diameter). The calculation of nutrient balances was done by subtracting the outputs (nutrient leaching and crop uptake) from the inputs (from irrigation, rain and fertilization).

$$
\begin{aligned}
\text{Balance} \,(\text{kg ha}^{-1}) \\
= \text{inputs (nutrients in fertilizer and irrigation water) } (\text{kg ha}^{-1}) \\
- \text{crop uptake } (\text{kg ha}^{-1}) - \text{ leaching } (\text{kg ha}^{-1})
\end{aligned}
$$

Gaseous emissions of nitrogen were not taken into account. Due to a lack of data for crop uptake of Na, this element was not incorporated in the balances. For testing of treatment effects on seasonal leachate volumes, cumulative element leaching and seasonal element balances a mixed model was applied using the *nlme* package of the R software (R Development core team 2008). The random factors used in the analyses were block and irrigation water quality. Fertilization and biochar application were used as fixed factors. In addition, we checked for statistically significant interactions between the factors. The full models were simplified stepwise until only significant factors ($p \leq 0.05$) remained. We checked for normal distribution of residuals (Shapiro–Wilk test) and heteroscedasticity of the data. If necessary, data were transformed by methods shown in the p value tables (Supplementary

material). Furthermore, the Pearson correlation coefficient was used to determine relationships between leaching components and differences between correlation coefficients were checked for significance using Fisher's Z transformation.

Results

Rainfall and irrigation amounts

During the first wet season, rainfall was 614 mm and irrigation 743 mm, resulting in total water input of 1357 mm (Table 1). In the following dry season, we recorded 48 mm of rainfall and 850 mm irrigation, amounting to a total input of 898 mm. The following wet season was rather dry with only 409 mm rainfall and only 408 mm of irrigation water was applied, amounting to a total input of 817 mm. In the last dry season, 140 and 1046 mm of rain and irrigation water were recorded, respectively, yielding total water input of 1186 mm.

Nutrient inputs with rainfall, irrigation water and fertilizer application

Mean pH values of wastewater were slightly higher in the wet seasons (8.15 and 7.46) compared to the dry seasons (7.31 and 7.13; Table 1). The same pattern was observed for tap water with pH values from 7.1 to 8.1. Rainwater had a pH between 6.9 and 7.5. Electrical conductivity was between 40.5 and 56.3 S m^{-1} in wastewater for all seasons with no clear pattern between wet and dry seasons. N inputs with wastewater were higher in the dry seasons (260 and 397 kg ha^{-1}) compared to the wet seasons (103 and 195 kg ha^{-1}), mainly due to the higher irrigation rates. N inputs with tap water irrigation (2–6 kg ha^{-1}) and with rainfall (0.2–1 kg ha^{-1}) were much lower. The same seasonal patterns were observed for P, K, Mg, Ca and Na inputs with wastewater irrigation and tap water and rain also showed smaller values for these parameters. Seasonal nutrient inputs into the system under FP treatments varied with crops and were between 118–311 kg N ha^{-1}, 29–98 kg P ha^{-1},

Table 1 Seasonal water and nutrient inputs into the multi-factorial field experiment in Tamale, Northern Ghana

	Amount (mm)	pH	EC (S m^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	K (kg ha^{-1})	Mg (kg ha^{-1})	Ca (kg ha^{-1})	Na (kg ha^{-1})
Wastewater									
Wet season 14	743	8.15	40.5	103	16	26	40	146	115
Dry season 14/15	850	7.31	56.3	260	95	36	49	221	161
Wet season 15	409	7.46	51.1	195	30	21	26	117	76
Dry season 15/16	1046	7.13	47.2	397	165	154	66	357	248
Tap water									
Wet season 14	743	7.35	19.1	2	< 1	8	17	70	15
Dry season 14/15	850	7.10	10.5	6	< 1	10	22	84	18
Wet season 15	409	8.10	8.9	2	< 1	4	10	40	7
Dry season 15/16	1046	7.10	9.8	5	1	23	28	113	32
Rain water									
Wet season 14	614	7.39	1.7	1	< 1	< 1	< 1	4	1
Dry season 14/15	NA								
Wet season 15	409	7.52	2.8	1	< 1	2	2	8	1
Dry season 15/16	140	6.90	2.8	< 1	< 1	1	1	9	1
Fertilizer									
Wet season 14				229	98	142	< 1	42	9
Dry season 14/15				118	50	73	< 1	22	4
Wet season 15				311	29	37	< 1	14	3
Dry season 15/16				149	56	73	< 1	27	6

NA no data available due to lack of rain

38–142 kg K ha^{-1},   0.1–0.3 kg Mg ha^{-1}, 14–42 kg Ca ha^{-1} and 3–9 kg Na ha^{-1} (Table 1), respectively.

Leachate amounts

During the wet season '14, leached water reached a mean of 770 mm, which was the highest of all seasons (Fig. 1), amounting to about 57% of the total water inputs. No significant treatment effects were found. Starting in the dry season '14/15, leaching amounts were significantly higher with wastewater irrigation than with tap water ($p < 0.001$), with 274 and 173 mm, respectively. This was corresponding to 30 and 19% of the water inputs. Similar differences were observed for wet season '15 with 318 mm of leaching under tap water and 419 mm under wastewater irrigation, corresponding to 36 and 51% of the water inputs. Due to a loss of irrigation protocols, leaching rates under wastewater irrigation during the dry season '15/16 could not be calculated. Under tap water, irrigation there was 239 mm leachate in the second dry season, corresponding to 19% of the water inputs.

Seasonal leaching of NO$_3$-N, P, K, Ca, Mg and Na

During the first wet season, nitrate leaching was very high in the fertilized plots reaching up to 222 kg NO$_3$-N ha^{-1} (Fig. 2). Similar significant treatment patterns were also observed during the following seasons but with lower NO$_3$-N leaching rates (Supplements Tab. S 1). In addition, during dry season '14/15 and wet season '15 irrigation with waste water increased NO$_3$-N leaching compared to tap water irrigation. A significant influence of the biochar applications on NO$_3$-N leaching was not observed. N-leaching as NH$_4^+$-N was much lower compared to NO$_3$-N and generally below one kg ha^{-1} (data not shown). Furthermore, P leaching was also very low (< 1 kg ha^{-1}) except during the second wet season, when P leaching under waste water irrigation reached up to 2 kg ha^{-1} and was significantly higher than under tap water irrigation. Seasonal K leaching ranged from 2 to 4 kg ha^{-1} with much lower values during the dry season '14/15. Highest leaching rates were observed for Ca with up to 375 kg Ca ha^{-1} in Wet 1 and between 20 and 200 kg Ca ha^{-1} in the following seasons, with generally highest values in the FP treatments with mineral NPK fertilization. In the dry season '14/15 waste water irrigation also significantly increased the Ca leaching. Magnesium showed a very similar pattern like Ca but with lower values (Fig. 2). Sodium leaching was higher in the wet seasons with 13–140 kg Na ha^{-1} than in the dry season with 4–48 kg Na ha^{-1}. In seasons one to three a significant increase of Na leaching under waste water irrigation was found (Fig. 2 and supplementary material Tab. S1).

The application of biochar to the soil increased only P leaching in the wet season '15 significantly. No other effects of biochar on nutrient leaching or on water flux in any of the monitored nutrients or seasons could be found.
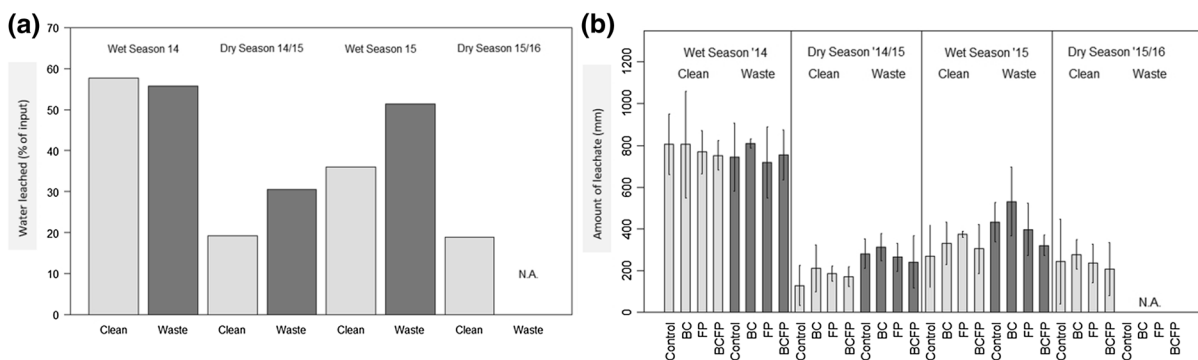

Fig. 1 Seasonal water flux (**b**) and the percentage of leaching to total water input (**a**) of tap (clean) water (light gray bars) and wastewater (dark grey) irrigated plots of the multi-factorial field experiment in Tamale, Northern Ghana. *Control* treatment (no additions), *BC* biochar treated soil, *FP* fertilization according to farmer's practice, *BCFB* fertilization according to farmers practice and biochar treatment. Error bars represent the standard deviation of means
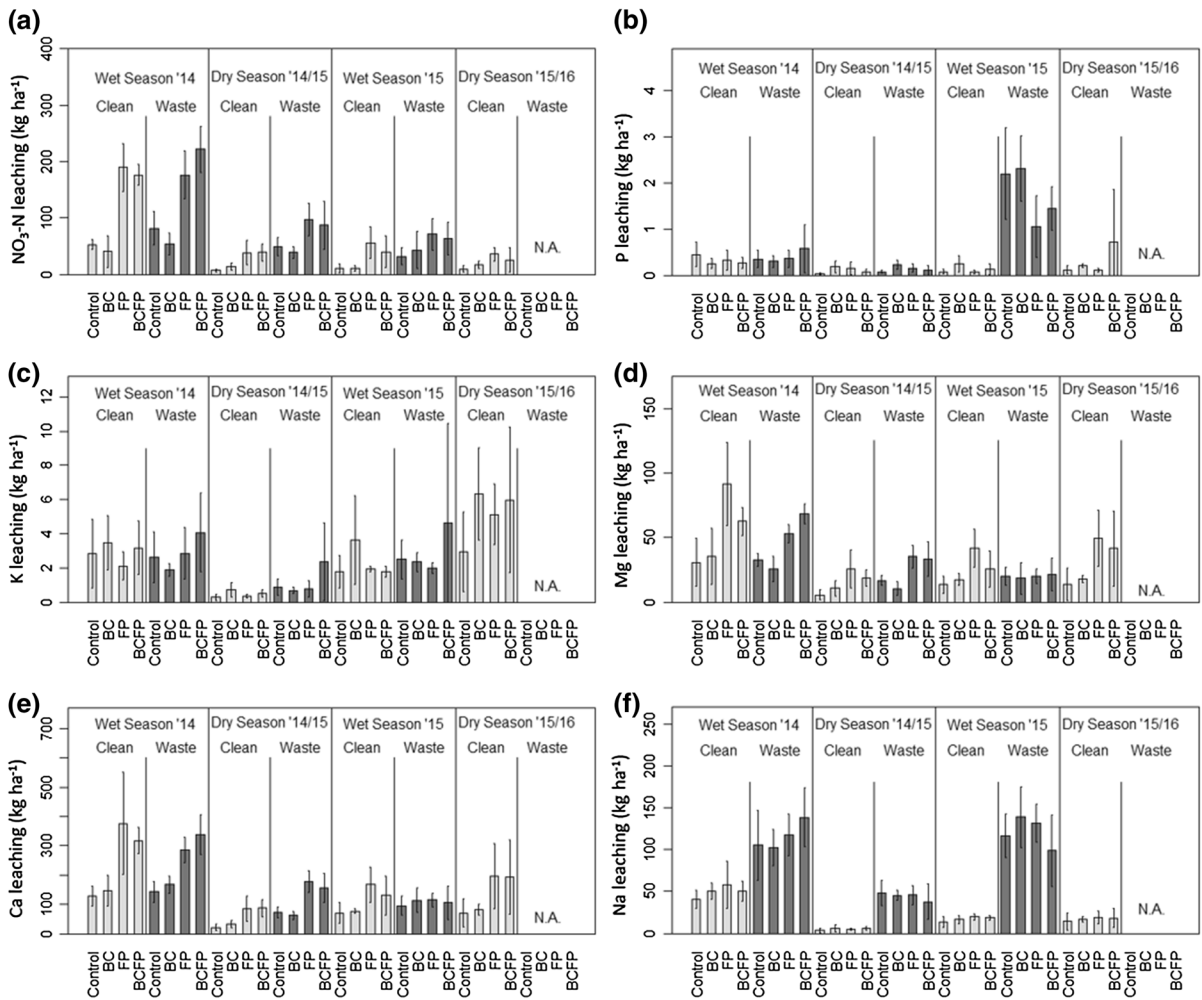
Fig. 2 Seasonal leaching of NO$_3$-N (**a**), P (**b**), K (**c**), Mg (**d**), Ca (**e**) and Na (**f**) under tap (clean) water (light gray bars) and wastewater (dark grey) irrigated plots of the multi-factorial field experiment in Tamale, Northern Ghana. *Control* treatment (no additions), *BC* biochar treated soil, *FP* fertilization according to farmer's practice, *BCFB* fertilization according to farmers practice and biochar treatment. Error bars represent the standard deviation of means, n = 4

Seasonal nutrient balances and contribution of leaching to total system output

Nutrient balances, in this study, were calculated to assess the sustainability of the system. Negative balances indicated a loss of nutrients from the system and thus a decline in soil fertility over time due to reduced nutrient stocks. When balances are positive, plant nutrients are accumulating in soil and fertility is build up. Neutral balances showing equilibrium between input and output of the system.

Nitrogen balances for the first wet season were negative to neutral and the data showed a non-significant effect of water quality to more negative balances under tap water irrigation (Fig. 3 and supplementary material Tab. S 2). Only during this season, leaching was the dominant N output from the soil system (53–83% of total N output) and exceeded the N uptake of plants (Fig. 4). During the following seasons, N uptake by plants was more dominant and leaching accounted for 11–32% of the total N removal form the system. In the first dry season tap water irrigated treatments also showed negative balances up to − 100 kg N ha^{-1} in the fertilized treatments. Under wastewater irrigation with its associated higher N-inputs (Table 1), N balances were positive with a surplus of up to + 65 kg N ha^{-1} in unfertilized treatments while fertilized treatments showed neutral
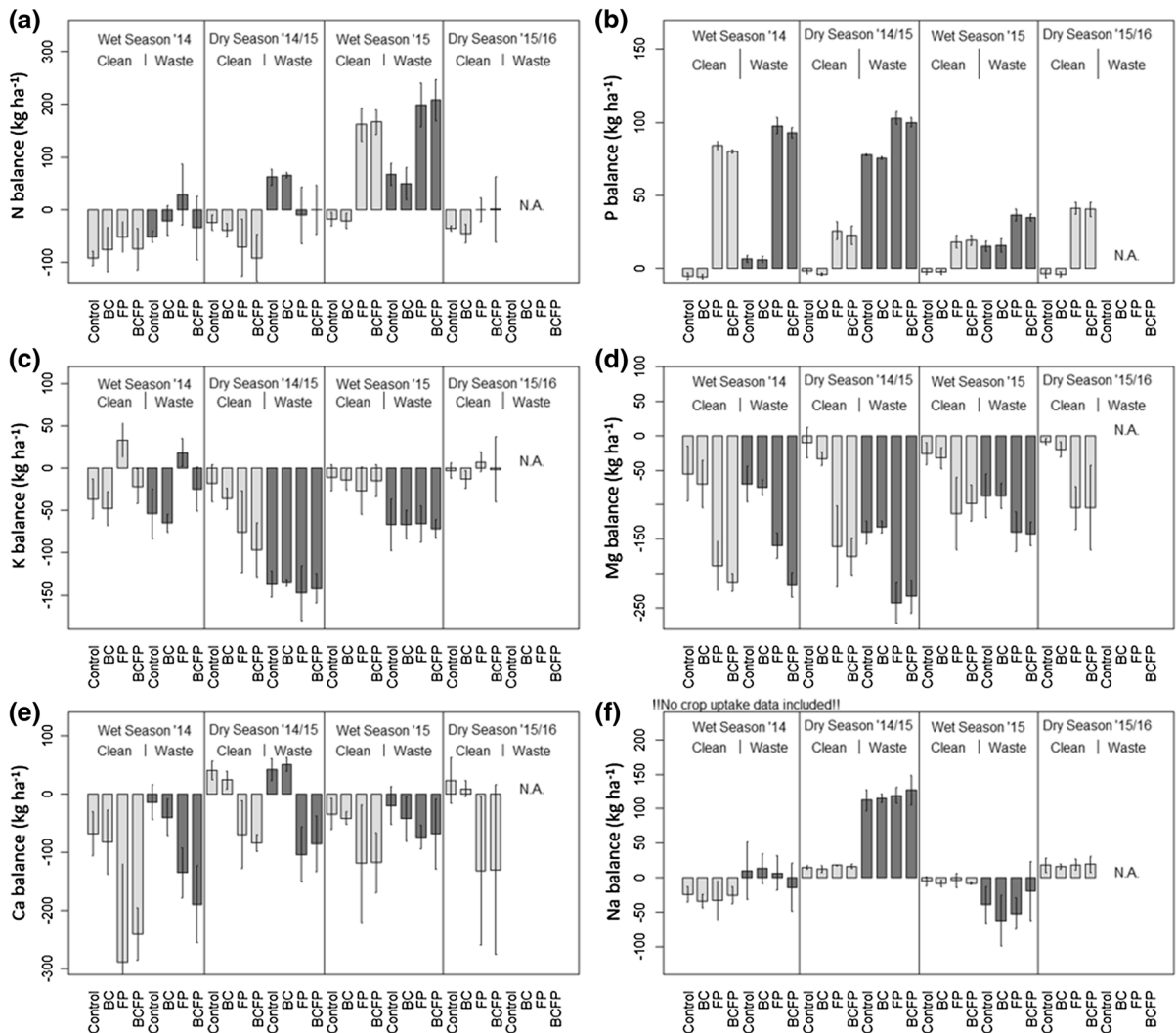
Fig. 3 Seasonal balances of N (**a**), P (**b**), K (**c**), Mg (**d**), Ca (**e**), Na (**f**) under tap (clean) water (light gray bars) and wastewater (dark grey) irrigated plots of the multi-factorial field experiment in Tamale, Northern Ghana. *Control* treatment (no additions), *BC* biochar treated soil, *FP* fertilization according to farmer's practice, *BCFB* fertilization according to farmers practice and biochar treatment. Error bars represent the standard deviation of means, n = 4. The Na balance was calculated without uptake by the crops because of missing data

N balances. During the wet season '15, the N balances for all the fertilized plots, especially under waste water irrigation were the highest with values between 166 and 209 kg N ha^{-1}, thus showing that much of the N inputs from fertilization (310 kg ha^{-1}) and waste water (195 kg ha^{-1}) were neither leached nor taken up by the crops. In the second dry season, N balances were only available for the tap water irrigated treatments and were negative to neutral (− 46 to 14 kg N ha^{-1}).

Phosphorus balances of plots with tap water irrigation without mineral fertilizer application were generally slightly negative with values between − 2 and − 6 kg P ha^{-1} (Fig. 3), largely due to crop uptake since P leaching is almost negligible (Fig. 1). With fertilizer application, balances were positive reaching up to 103 kg P ha^{-1} (Fig. 3). In the dry season '14/15 and the following wet season waste water application without fertilizer caused positive balances of 78 and 13 kg ha^{-1}, respectively. In all seasons, P balances were significantly increased by fertilizer application
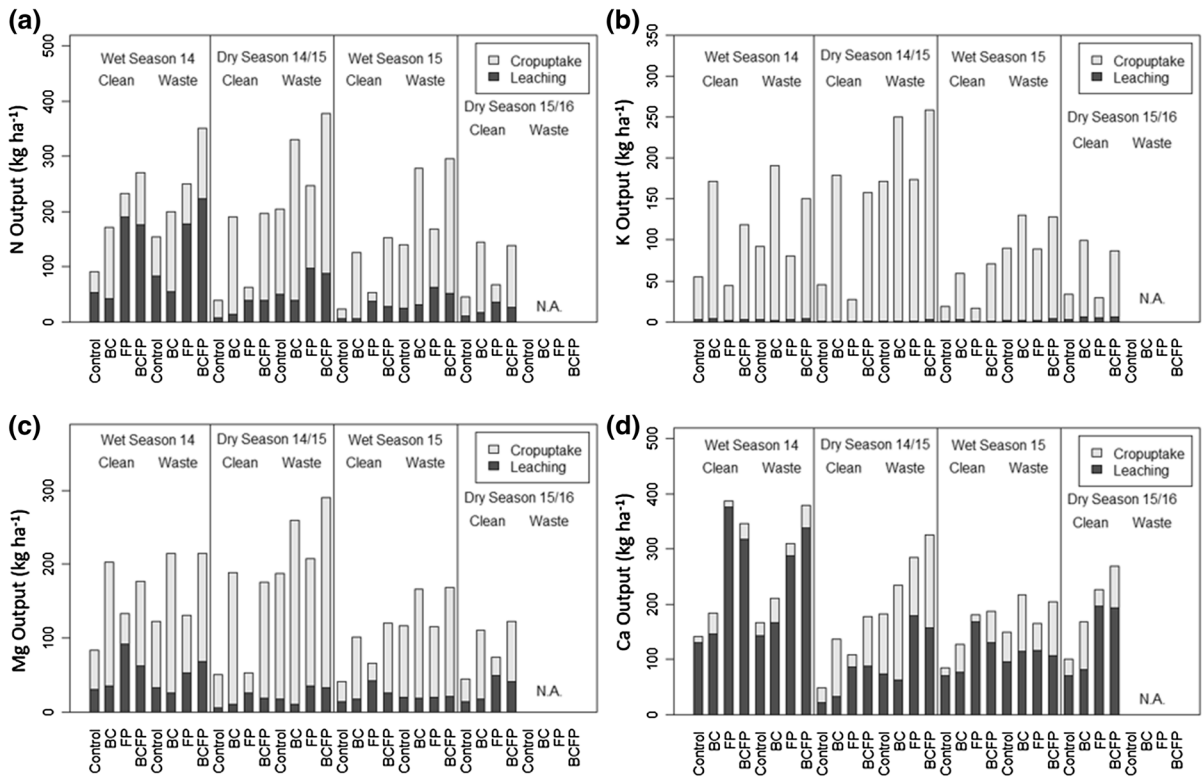
Fig. 4 Contribution of leaching and crop uptake to the total output of N (**a**), K (**b**), Mg (**c**) and Ca (**d**) under tab (clean) and wastewater irrigation in the multi-factorial field experiment in Tamale, Northern Ghana. *Control* treatment (no additions), *BC* biochar treated soil, *FP* fertilization according to farmer's practice, *BCFB* fertilization according to farmers practice and biochar treatment

and during the first three seasons, a significant effect of wastewater irrigation was determined (Supplementary material Table 2). The contribution of P leaching to total P output was generally low (Fig. 4). Even with the elevated P leaching under wastewater irrigation in the wet season '15, leaching accounted only for 2% of the total P output.

Potassium balances were generally negative, except for the NAP treatments during the first wet season (Fig. 3) when K fertilizer application was highest (Table 1). Similar to P, K removal occurred mainly with crop biomass and leaching losses were negligible (Fig. 4). Interestingly, during the dry season '14/15 and the following wet season, K balances were more negative under waste water irrigation, because of increased uptake of K by the crops, although inputs were higher by about 36 and 21 kg ha^{-1} compared to those under tap water irrigation (Table 1).

Magnesium balances were negative in all seasons and treatments, with highest losses in the FP treatments where NPK fertilizer was applied (Fig. 3). Here, the higher outputs were generally due to both increased leaching and higher crop uptake (Fig. 4). Interestingly, the contribution of Mg leaching to total system output was higher under tap water than under wastewater irrigation in the wet seasons.

Calcium balances were also mostly negative except in the unfertilized plots during the two dry seasons (Fig. 3). Outputs were dominated by leaching and this was greatly enhanced by fertilizer application (Fig. 4). In the first season, Ca leaching accounted for 87–94% of total Ca output. In the following season, this contribution was lower but still dominant. The lowest contribution was recorded in the dry season '14/15 with 37–55%. Due to higher Ca input with wastewater irrigation, Ca balances were higher. However, the effect of irrigation water quality was not significant.

Sodium balances in the first wet season were between − 35 and 12 kg Na ha^{-1} and showed a significant increase by wastewater irrigation. This is in

line with the higher Na input (100 kg Na ha^{-1}) with wastewater irrigation. During the dry season '14/15 wastewater irrigation showed the biggest effect and increased balances to 127 kg Na ha^{-1}. Surprisingly, during the second wet season wastewater irrigation decreased Na balances to − 61 kg Na ha^{-1}, while tap water irrigated plot exhibited almost neutral balances. This is likely connected to the increased Na leaching rates and water flux under wastewater irrigation. When compiling a partial Na balance for the first and second wet season, we found a negative mass balance under tap water irrigation with − 15 to − 30 kg Na ha^{-1} and under wastewater irrigation, a positive balance of 66–95 kg Na ha^{-1}.

Discussion

Range of nutrient inputs

Comparing N inputs of this study (702–1051 kg N ha^{-1} year^{-1} in fertilized and wastewater irrigated plots) with data from other studies on high input urban agriculture shows a relatively low nutrient input in our case. Diogo et al. (2010) reported inputs up to ∼ 1400 kg N ha^{-1} year^{-1}. In studies from Vietnam and China N inputs were up to 1643 and 1565 kg N ha^{-1} year^{-1}, respectively (Huang et al. 2006; Khai et al. 2007). Graefe et al. (2008) reported even annual inputs of up to 7519 kg N ha^{-1}. However, the N inputs in our study were higher than the reported up to 165 kg N ha^{-1} year^{-1} from Goenster et al. (2015) for home gardens in the Nuba Mountains of Sudan or 800 kg N ha^{-1} year^{-1} reported by Abdalla et al. (2012) also for a study in River Nile flooded gardens of Khartoum, Sudan.

The P inputs in this study are between 159 and 279 kg P ha^{-1} year^{-1} with fertilizer and wastewater use. As in the case of N inputs, this is lower than 500, 261, 736 and 543 kg P ha^{-1} year^{-1} reported by Graefe et al. (2008), Diogo et al. (2010), Khai et al. (2007) and Huang et al. (2006), respectively. However, the P applied in this study is higher than the inputs reported by Abdalla et al. (2012) and Goenster et al. (2015) with 4.5 and 29 kg P ha^{-1} year^{-1}, respectively.

K inputs in this study were as high as 276 and 284 kg K ha^{-1} year^{-1} and the comparison with other reported amounts show the same picture as for N and P. K inputs were much lower than values reported by Diogo et al. (2010), Khai et al. (2007) or Huang et al. (2006) of 700, 903 or 690 kg K ha^{-1} year^{-1}, respectively. Interestingly, Graefe et al. (2008) reported a K input of up to 7081 kg K ha^{-1} year^{-1}. However, Abdalla et al. (2012) and Goenster et al. (2015) reported lower values with 90 and 149 kg K ha^{-1} year^{-1}.

Irrigation water use and effects of waste water on drainage

The percentage of leached water was very high in the first season. Although it was a wet season, the irrigation was high with a total water input of 1356 mm and 56% of the total water input was leached. During the following season, the total amount of irrigation and rainfall was lower and therefore the leaching rates were reduced and the use of irrigation water was improved. In another study, Sangare et al. (2012) measured water lost by drainage of about 3–8% of total water input in urban farm in Bobo Dioulasso, Burkina Faso. These amounts were exceeded in our study in all seasons, indicating an inappropriate irrigation management.

Furthermore, during the dry season '14/15 and the following wet season we observed increased water fluxes under wastewater irrigated plots. This could maybe be explained by high loads of humic matter, polysaccharides and fatty acids in waste water causing increasing soil water repellency, resulting in higher infiltration rates as shown by Arye et al. (2011). However, Schacht and Marschner (2015) reported lower hydraulic conductivity when soil was irrigated with treated waste water compared to fresh water in fruit orchards in Israel. The irrigation with waste water and the resulting soil water repellency can cause finger flow paths in soil with higher water flux in these areas (Wallach et al. 2005). In addition, damage of macro and micro pore structure under irrigation with treated sewage effluent has been observed and the losses of Ca may further increase this effect (Gonçalves et al. 2007). The thereby reduced pore space may reduce the water holding capacity of the soil. This may explain the increased leachate amounts under wastewater irrigation.

An effect of biochar on leachate amounts was not found in our study. A literature review reveals inconsistent effects of biochar on soil physical

parameters. Biochar showed decreasing and increasing effects on saturated hydraulic conductivity depending on biochar particle size and soil texture (Lim et al. 2016). However, biochar additions had no effect in loamy soils. In other studies, biochar increased the water holding capacity of soil (Abel et al. 2013; Sun and Lu 2014). In a study of Ojeda et al. (2015), biochar had no effect on water storage in soil. In general, biochar properties and their effect on soil can vary widely depending on feedstock material and pyrolysis method.

Seasonal nutrient leaching effected by biochar

In this study, we did not find any significant effects of biochar on leaching of nutrients. There are some field studies on nutrient leaching and biochar application in the literature. In a Colombian Oxisol, leaching of N, Ca, Mg and K decreased after the addition of 20 t ha^{-1} wood biochar (Major et al. 2012). Also N leaching from soil was reduced by 82% in a field trial in New York with the application of biochar produced from maize stover under full fertilization but not with half fertilization (Güereña et al. 2013). Hardie et al. (2015) conducted a wick lysimeter campaign in a young apple orchard in New Zealand with 47 t ha^{-1} wood biochar which had no effect on nitrate leaching but increased K losses. The lack of effects in our study may be explained with the high inputs of nutrients through fertilizer application and wastewater irrigation in this vegetable production system and the properties of the biochar used in this study. The surface area of the used rice husk char is small compared to other biochars (Mukherjee et al. 2011), therefore the interface of biochar to interact with soil water and sorption of substances is limited. Furthermore, the application rate of 20 t ha^{-1} incorporated to a depth of 0.2 m or about 0.5 wt% is relatively low but realistic. Many other biochar studies used biochar additions which were ten times higher or more to observe significant effects of biochar on leaching.

Wastewater effects on seasonal nutrient leaching

The seasonal N leaching was effected by wastewater irrigation. This is mainly explained by the increased N input through wastewater. In the first season, due to the high rainwater input and therefore high water flux in the soil profile, the effect of wastewater irrigation on N leaching was less pronounced compared to the later seasons. Also Barton et al. (2005) found an increase of N leaching when soils were irrigated with secondary treated domestic effluent. In our study, P leaching was generally very low with less than 1 kg ha^{-1} per season because the soil exhibits many bonding sides for P like Al and Fe hydrous oxides and clay (Parfitt 1979). Nevertheless, in the wet season '15 an increase of P leaching under wastewater irrigated plots has been observed. This could be explained by competitive sorption of waste water borne dissolved organic carbon with phosphorus as shown by Kang et al. (2011) or more specifically organic anions (Kafkafi et al. 1988). This mechanism in combination with an increase of soil pore water velocity, due to the above mentioned effects of waste water irrigation on soil water repellency and physical soil properties could likely have cause the measured increment in P leaching in the wet season '15.

Furthermore, K leaching was only increased in the dry season '14/15 by waste water irrigation, which is explained by the negative K balances of the systems. Also in the literature, K leaching is negligible in these savannah soils (Piéri 1992). The leaching of Ca and Mg was also observed to be increased by waste water irrigation. This is attributed to the increased nitrate leaching because an equivalent amount of cations is needed to obtain electrical neutrality of the leachate (Lehmann and Schroth 2003). This is also supported by higher Pearson correlation coefficients of leached Ca and Mg to leached NO_3-N (0.65 and 0.73, $p < 0.05$) than to the water flux (0.55 and 0.6, $p < 0.05$). The difference between correlation coefficients were found to be significant ($p < 0.01$, Fisher Z). When compared to other studies, N leaching was much higher in the first season under fertilizer application than reported 2.2–7.3 kg ha^{-1} year^{-1} by Predotova et al. (2011) or Safi et al. (2011) who found an average N leaching of 140 kg N ha^{-1} year^{-1} in two vegetable gardens in Kabul, Afghanistan and the reported 53–45 kg N ha^{-1} year^{-1} by Goenster et al. (2015). In season two to three, the reported values are close to ours with the exemptions of Predotova et al. (2011), who reported much lower leaching rates which, however, are likely reflecting the particularities of a below average rainfall year. Our findings indicate that apart from the effects of soil properties and input quantities, leaching is largely controlled by the water flux. The aforementioned studies were done by using

ion exchange resins cups and therefore water fluxes could not been measured (Weihermüller et al. 2007). The leaching of P was as low as reported in other studies (Goenster et al. 2015; Predotova et al. 2011) but lower than the 6.5 kg ha^{-1} year^{-1} from the study of Safi et al. (2011) which was conducted in a garden soil with a pH higher than 8 and maybe therefore higher P mobility, since P mobility is strongly effected by soil pH (Arai and Sparks 2007). Furthermore, K leaching was far lower than 33–34 kg ha^{-1} year^{-1} reported by Goenster et al. (2015).

When we compare the N output from the system by crop uptake with leaching, especially when water fluxes are high like in the first wet season, leaching can be the major output pathway. In later seasons with are more adjusted irrigation and therefore less water flux, the contribution of leaching to the total output was about 25% (Fig. 4). In the case of P and K which are less prone to leaching, the leaching losses are almost negligible.

Nutrient balances in urban agriculture

Nitrogen mass balances were negative in the first two seasons but positive in the third season. Generally, the results are comparable to those reported by Cobo et al. (2010) in their meta-analysis. Wastewater borne N contributed strongly to a positive balance although higher water fluxes were observed in the later seasons. A general conclusion whether N balances are negative or positive cannot be given but underline the high variability in this system. However, the N balances in this study were far lower than the extremely high horizontal balances [Horizontal balance (kg ha^{-1}) = inputs (kg ha^{-1}) – crop uptake (kg ha^{-1})] of 7312 (Graefe et al. 2008), 1133 (Diogo et al. 2010), 2056 (Lompo et al. 2012) or 814 kg N ha^{-1} year^{-1} (Huang et al. 2006). Nevertheless, they are comparable to N balances of − 70 kg N ha^{-1} year^{-1} reported by Goenster et al. (2015) or Abdalla et al. (2012) with 342 kg N ha^{-1} year^{-1}. The loss of N from the soil during the experiment was also measured in the soil parameters. The initial low soil N of 0.04% (0–0.2 m) and 0.03% (0.2–0.4 m) was further reduced by 3.5–10.1% (Häring et al. 2017).

Strongly positive balances for P were observed in this study, which was comparable to the balances calculated by other researchers (Diogo et al. 2010; Graefe et al. 2008; Huang et al. 2006; Khai et al.

2007). Negative balances as reported by Goenster et al. (2015) or Abdalla et al. (2012) seems to be impossible with the soil in this study when fertilizer is applied to soil and/or waste water is used for irrigation. A completely different situation was found for K with negative balances. The use of wastewater was decreasing the balances although higher K inputs occurred. This is explained by higher K plant uptake as measured by Akoto et al. (2018). Mg and Ca showed negative balances and are therefore strongly removed from the soil. In addition, fertilization caused a significant reduction of Mg and Ca balances. The leaching of these basic cations decreases the acid buffering capacity of the soil and thus indicates soil acidification. This was confirmed by significantly lower soil pH values in treatments where fertilizer was used (Häring et al. 2017). Furthermore, effects of wastewater irrigation were reflected in positive Na balances, while negative balances were found under tap water irrigation. Therefore, Na was accumulating in soils under waste water irrigation which might have increased soil salinity and the exchangeable sodium percentage (Levy et al. 2014). Interestingly, Na balances under wastewater irrigation were not always positive. During wet seasons, balances close to zero or even negative were found so that salinization during dry season may be reversed during the rainy season.

Conclusions

Unlike rural agricultural systems, African urban agricultural systems are typically highly intensive input–output systems that show large positive nutrient balances for N and P. Waste water contributes significantly to the nutrient inputs as shown in this field trial. Nutrient leaching and nutrient mass balances are also affected by waste water irrigation, the amount of fertilizer applied and strongly by the appropriate amount of irrigation. In our case, K fertilization was not high enough to replace K losses by plant uptake. There also were high net losses of basic cations, indicating long-term acid buffering substances, especially with mineral fertilizers. This should be avoided by better fertilizer management and compensated by inputs of buffering agents, possibly with organic amendments such as composts.

Our results and field observations show that there is scope for better fertilizer management taking waste

water contribution into account and adjusting irrigation quantities. Apart from the nutrient contribution from waste water there are also negative effects on soil physical parameters that result in higher leachate water losses and an increased salinization risk from Na accumulation in the soils. Contrary to other studies, effects of biochar additions on any of the studied parameters were not observed.

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