Received: 17 December 2019 | Accepted: 14 May 2020
DOI: 10.1111/gcbb.12719

ORIGINAL RESEARCH

GCB-BIOENERGY BIOPRODUCTS FOR A SUSTAINABLE BIOECONOMY Open Access | WILEY

Comparison of maize, permanent cup plant and a perennial grass mixture with regard to soil and water protection

Dennis Grunwald[1] | **Kerstin Panten**[1] | **Andreas Schwarz**[2] | **Wolf-Anno Bischoff**[2] | **Siegfried Schittenhelm**[1]

[1]Institute for Crop and Soil Science, Julius Kühn Institute (JKI), Braunschweig, Germany
[2]TerrAquat Consultants, Nürtingen, Germany

Correspondence
Dennis Grunwald, Institute for Crop and Soil Science, Julius Kühn Institute, Bundesallee 58, 38116 Braunschweig, Germany.
Email: dennis.grunwald@julius-kuehn.de

Funding information
This study was supported by the German Federal Ministry of Food and Agriculture via the Agency for Renewable Resources under grant number 22023914.

Abstract

Agricultural production of biogas maize (*Zea mays* L.) causes hazards to aquatic ecosystems through high levels of nitrogen (N) inputs. Newly introduced and already established perennial crops such as the cup plant (*Silphium perfoliatum* L.) and perennial grass mixtures offer the possibility of more environmentally friendly agricultural bioenergy production. The objectives of this field study were to quantify and compare soil mineral N, water infiltration, water runoff, soil erosion and N leaching under maize, permanent cup plant, and a perennial grass mixture. The study was conducted from October 2016 to March 2019 in Braunschweig, Germany. Plots with cup plant and grass mixture exhibited lower mineral N contents than maize, especially between 30 and 90 cm soil depth. Soil water infiltration was significantly different between the three crops. The grass mixture had the highest infiltration rates (6.2 mm/min averaged across 3 years), followed by cup plant (3.6 mm/min) and maize (0.9 mm/min). During wet periods, higher N leaching was found for maize (up to 42 kg N ha^{-1} $year^{-1}$) than for cup plant (up to 5 kg N ha^{-1} $year^{-1}$) or the grass mixture (up to 11 kg N ha^{-1} $year^{-1}$). While runoff and erosion for cup plant and the grass mixture were negligible during the study period, considerable amounts of runoff water and eroded sediment of up to 1.5 Mg ha^{-1} $year^{-1}$ were collected from the maize plots despite the near flat terrain of the experimental field. Overall, permanent cup plant proved suitable as a component for energy cropping systems to reduce the risk of N leaching and soil erosion, which is particularly important for the preventive flood protection in view of the more frequent occurrence of high intensity rainfall under climate change conditions.

KEYWORDS
cup plant, maize, mixed perennial grasses, nitrogen leaching, perennial crops, soil erosion, water runoff

1 | INTRODUCTION

High levels of nitrogen (N) inputs into surface and groundwater bodies through runoff and leaching from agricultural areas remain an important environmental issue on intensively used farmland worldwide. Nitrate levels exceeding 50 mg/L are found in 28% of the German groundwater bodies (BMU [Federal Ministry of the Environment, Nature Conservation, and Nuclear Safety] & BMELV [Federal Ministry of Food, Agriculture and Consumer Protection], 2016). Depending on the soil texture, high levels of N losses are associated with annual crops such as maize (*Zea mays* L.; Zhou & Butterbach-Bahl, 2014) with particularly high losses on well-drained sandy soils (de Ruijter, Boumans, Smit, & van den Berge, 2007; Köhler, Duynisveld, & Böttcher, 2006). With an area of 900,000 ha, accounting for about 36% of the total maize area (FNR, 2018), biogas maize is by far the most important biogas crop in Germany. When maize is grown conventionally, that is, with winter fallow and ploughing before sowing on slopes or soils with low water infiltration, the risk of soil and nutrient losses due to water erosion increases significantly (Palmer & Smith, 2013; Vogel, Deumlich, & Kaupenjohann, 2016). This problem is further aggravated through trends towards higher precipitation intensities in spring and winter (Zolina et al., 2008) and higher rainfall erosivity in early summer (Fiener, Neuhaus, & Botschek, 2013), when ploughed maize fields are most vulnerable because of the freshly tilled soil and the lack of plant cover.

Grass silage, for example from a perennial grass mixture, is the second most important biogas substrate in Germany after maize with a share of 14% (FNR, 2018). Soils under perennial grassland were found to have higher abundances or activity of earthworms than maize (Lamandé, Hallaire, Curmi, Pérès, & Cluzeau, 2003; Whalen, 2004), which might lead to a higher water infiltration rate and reduced water erosion (Blouin et al., 2013; Fischer et al., 2014). However, biomass yields from grasslands are only about half of maize (Weiland, 2010). Cup plant (*Silphium perfoliatum* L.) is a promising new second-generation perennial bioenergy crop with a high methane yield potential (Haag, Nägele, Reiss, Biertümpfel, & Oechnser, 2015; Mast et al., 2014). Cup plant has shown increased earthworm abundances in comparison to maize as well (Emmerling, 2014; Schorpp & Schrader, 2016), and thus can also be expected to increase water infiltration and reduce surface water runoff and erosion. In addition, the higher evapotranspiration and deeper rooting of permanent cup plant compared to annual maize (Schoo, Schroetter, Kage, & Schittenhelm, 2017; Schoo, Wittich, Böttcher, Kage, & Schittenhelm, 2017) can be expected to cause a higher capacity for water storage and to reduce deep drainage and therefore N leaching. Cup plant continues to grow after the harvest in autumn, thereby taking up N and preventing it from leaching during the winter without the need to establish a catch crop after harvest. Because of its ecological benefits, since 2018 farmers are receiving direct payment for growing cup plant under the EU greening programme (EUR-Lex, 2018).

Previous studies found that perennial crops offer new opportunities for sustainable agricultural bioenergy production. *Miscanthus* (*Miscanthus* × *giganteus* Greef & Deuter) and switchgrass (*Panicum virgatum* L.), for example, were shown to reduce N leaching compared to annual crops (Ferchaud & Mary, 2016; Hussain, Bhardwaj, Basso, Robertson, & Hamilton, 2019; McIsaac, David, & Mitchell, 2010; Pugesgaard, Schelde, Larsen, Lærke, & Jørgensen, 2015). This beneficial effect is mainly related to the longer growing period and hence the increased opportunity for N uptake, lower seepage through higher evapotranspiration and less mineralization due to the absence of tillage operations (Pugesgaard et al., 2015). The lack of tillage in perennial compared to annual crops in a Mediterranean environment considerably reduced soil erosion on slopes (Cosentino, Copani, Scalici, Scordia, & Testa, 2015). In contrast to maize, perennials were also found to increase the soil organic matter content (Gauder et al., 2016), which may improve the soil structure. Routschek, Schmidt, Enke, and Deutschlaender (2014) noted that changes in land use or crop management such as conversion of arable land to pasture and the change from conventional to conservation tillage has the potential to counterbalance the higher risk of erosion resulting from changed precipitation patterns in a future climate.

This study analysed water and N fluxes in maize, cup plant and a perennial grass mixture over two and a half years in a field trial in the North German Plain. Soil moisture, soil mineral N contents, water infiltration, N leaching as well as water runoff and soil erosion were measured. The study aimed to test the following hypotheses: (1) higher infiltration rates under cup plant and grass significantly reduce surface runoff and water erosion compared to maize, (2) cup plant and grass pose a lower risk of groundwater contamination by N than maize due to a lower soil mineral nitrogen content and (3) after a few years of cultivation permanent cup plant achieves similar benefits to grass in terms of soil and water protection, as indicated by reductions of nutrient losses to other ecosystems, soil loss through erosion, flood risks through runoff and by a favourable N balance.

2 | MATERIALS AND METHODS

2.1 | Field experiment

2.1.1 | Site characteristics

The experiment was conducted between October 2016 and March 2019 on the experimental field (52.296°N, 10.438°E, altitude 76 m) of the Institute for Crop and Soil Science in Braunschweig, Germany. The terrain is very flat with a slope of only 1%–2%. The soil is classified as Haplic Luvisol with locally compacted clay-rich bandings in the subsoil typical of a Lamellic Luvisol (IUSS Working Group WRB, 2015). The soil texture varies with depth with 60% sand, 34% silt and 6% clay in the 40 cm topsoil, 78% sand, 17% silt and 6% clay at 40–80 cm soil depth and 92% sand, 4% silt and 4% clay at 80–120 cm soil depth. The site receives an average annual precipitation of 616 mm and the average daily temperature is 9.1°C.

2.1.2 | Experimental design

The field experiment was designed as two-factorial split-plot with four replications, two water regimes (with and without heavy rainfall simulation [HRS]) as main plots and three biogas crops (continuous maize, permanent cup plant and perennial grass mixture) as subplots. The subplots had a size of 240 m^2 (6 m × 40 m). Heavy summer rainfall was simulated by means of overhead irrigation with a travelling sprinkler (Hüdig GmbH & Co. KG) on four consecutive days at the end of July, with daily amounts of 20 mm within 60 min per plot in 2017 and 30 mm within 90 min per plot in 2018. The difference in irrigation amounts became necessary because of the strongly different weather conditions in both years. A daily amount of 30 mm precipitation corresponds to the maximum daily amount per year averaged over the last 15 years at the study site. The used hybrid maize cultivar 'Walterinio' (S 270) of KWS Saat SE is suitable for silage and biogas production. Cup plant seedlings of Russian origin were obtained from N.L. Chrestensen. The perennial grass mixture 'Gasindex 2401' from Deutsche Saatenveredelung AG consisted of 20% each of orchard grass (*Dactylis glomerata* L.), Festulolium (*Festuca* spec. × *Lolium* spec.), perennial rye grass (*Lolium perenne* L.), meadow fescue (*Festuca pratensis* Huds.) and tall fescue (*Festuca arundinacea* Schreb.).

2.1.3 | Crop management

At the beginning of the study period in autumn of 2016, maize, cup plant and the perennial grass mixture had already been grown on the same plots for five growing seasons, starting in 2012. It was only in 2015 that lucerne grass has been replaced with the perennial grass mixture (hereinafter referred to as grass) described in the previous section. The maize plots were ploughed to a depth of 25 cm before sowing each year and the seedbed was prepared with a rotary harrow. Maize was sown on 2 May 2016, 4 May 2017 and 3 May 2018. Maize density was 95,000 plants ha^{-1} (0.75 m row distance). The cup plant was initially spaced 0.5 m × 0.5 m (40,000 seedlings ha^{-1}). However, by the start of the experiment, the plants had spread widely on the plots and left little free soil surface. The grass was seeded at 40 kg/ha. The vegetation period started on 21 March 2016, 17 March 2017 and 12 April 2018. N fertilizer was applied in the form of calcium ammonium nitrate. Maize received 180 kg N/ha on each of 12 May 2016, 16 May 2017 and 16 May 2018, while cup plant received 170 kg N/ha on each of 23 March 2016, 28 March 2017 and 21 March 2018. A total of 280 kg N/ha in 2016 and 310 kg N/ha in each of 2017 and 2018 was applied to the grass by split application at the beginning of the growing season and after each of the first four harvests. Because of the heat and drought experienced throughout the summer of 2018, the whole experiment was irrigated by providing 30 mm of water within 3 hr per plot once a week for a 9-week period from mid-June to mid-August. Cup plant and maize were harvested on 2 September 2016, 12 September 2017 and 31 August 2018 with row-independent forage harvesters (CLAAS Jaguar 950 and KRONE BIG X) cutting the entire biomass of each plot in one single passage. The grass mixture was cut five to six times a year using a HALDRUP field plot harvester.

2.2 | Soil moisture monitoring

A Diviner 2000 soil moisture probe (Sentek Technologies) was used for monitoring the soil water content. One PVC access tube (Ø 60 mm) was installed in each of the 24 field plots. Soil moisture readings were taken at 10 cm intervals from 5 to 155 cm, twice a week during the growing season and once a week during the winter months. The Diviner 2000 scale frequency readings were converted to volumetric soil water content (vol.%) using site- and depth-specific calibration equations. The coefficient of determination between the vol.% and the gravimetrically determined soil water content was 0.64 (n = 1,662). Measurements in the cup plant and grass plots began in March 2017, while measurements in the maize plots began after ploughing in May 2017.

2.3 | Soil mineral N

Soil samples for mineral N (N_{min}) analyses were taken in October 2016, 2017 and 2018 and in March 2017, 2018 and 2019 at three depths (0–30, 30–60 and 60–90 cm) simultaneously to the installation or change of the Self-Integrating Accumulators (SIAs; see Section 2.6), as a mixed sample from six points per plot. Prior to analysis, samples were stored frozen. N in the form of nitrate (NO_3-N) and ammonium (NH_4-N) was analysed in 0.0125 M $CaCl_2$ extracts by continuous flow analysis using a San^{++} Automated Wet Chemistry

Analyser (Skalar Analytical). Results were calculated as total N_{min} (NO_3-N + NH_4-N) kg/ha at the respective soil depth.

2.4 | Infiltration measurements

Water infiltration rates were measured with hood infiltrometers (UGT; Schwärzel & Punzel, 2007) at near-saturated conditions around the beginning of the vegetation period in 2017, 2018 and 2019. At individually chosen time intervals, the infiltrated water was measured until a constant infiltration rate was reached. Measurements were conducted at six points per plot. It should be noted that saturated hydraulic conductivities measured with a hood infiltrometer are about 10 times larger than with disk infiltrometers placed on a sand contact layer (Angulo-Jaramillo, Bagarello, Iovino, & Lassabatere, 2016; Schwärzel & Punzel, 2007).

2.5 | Runoff and soil erosion

Runoff and soil erosion were measured for a partition of each plot, covering a quarter (1.5 m) of the plot width and the whole (40 m) of the plot in length. To collect the water and eroded soil, 90 L containers were embedded in the ground at the end of the HRS plots. To allow the transition of water and soil to the containers, an apron leading into the container was installed at the end of the plot. Metal sheets were pushed into the soil to cover the width of 1.5 m. For the maize plots, the containers were installed at the end of one of four tracks caused by traffic, representing the actual proportions on the plot. After each rainfall, containers were checked and emptied if necessary. During the HRS, the containers were continuously emptied.

2.6 | Nitrate and ammonium leaching

To detect leaching of NO_3-N and NH_4-N below the main root zone, SIAs (TerrAquat Consultants; patent no. 197 26 813) were used. The function of SIAs is based on the infinite sink principle as described by Bischoff (2007). Briefly, cylindrical polypropylene cartridges (10 cm in diameter and in height), open at the top and with a mesh on the bottom, were filled with a combination of sand and mixed-bed ion-exchange resin to trap the leaching cations and anions from the soil leachate flowing through the SIA. All ions were accumulated within the SIA over the investigation period and remained adsorbed by the resin until extraction. In each plot, three SIAs were installed in one installation pit. For the installation, horizontal access tunnels were dug under the undisturbed soil. The SIAs were placed at 1.2 m depth, measured at the upper end of the cartridge. After the installation, all tunnels and the pit were carefully and compactly closed to avoid disturbances of soil hydraulics. The SIAs were first installed in October 2016 and changed each March and October to cover the summer (mid-March to mid-October) and winter (mid-October to mid-March) leaching periods until March 2019. After removing the SIAs, the upper half, which is supposed to hold all leached nutrients, was thoroughly mixed. After extraction with 1 M NaCl, NO_3^--N and NH_4^+-N were determined colorimetrically with a Type Evolution II continuous flow analyser (Alliance Instruments). Total N leaching in kg N/ha was calculated by multiplication of the surface area of the SIA with the amount of leached N and then scaling up. N leaching in the following refers to the sum of NO_3^--N and NH_4^+-N.

2.7 | Biomass analysis and N uptake

After harvest, the fresh matter yield of the aboveground plant material per plot was determined. Two subsamples were dried at either 105°C to obtain dry matter yield or at 60°C for a subsequent analysis of total N contents with an ELEMENTAR vario MAX cube C/N analyser (ELEMENTAR). Total N contents were multiplied by dry matter yields to obtain the N uptake into the aboveground biomass. For the grass plots, this procedure was carried out for each harvest. Additionally, the regrowth of cup plant was analysed for its potential to act as cover crop taking up N after harvest in the autumn of 2019. To this end, two randomly chosen 1 m^2 subplots per cup plant plot were cleared of all cup plant residues shortly after harvest in the beginning of September 2019. Afterwards, the subsequent cup plant regrowth was harvested by hand on the same subplots in mid-November 2019. The plant material was then analysed for dry matter yield and N contents as described above.

2.8 | N balance

The annual N balances were calculated for each crop for the two periods mid-March 2017 to mid-March 2018 (2017–2018) and mid-March 2018 to mid-March 2019 (2018–2019). The N input was calculated using fertilizer data ($N_{fertilizer}$, see Section 2.1.3), N content in the irrigation water ($N_{irrigation}$) and the mean atmospheric N deposition ($N_{deposition}$). The value for the irrigation water represents the mean of the treatments with and without HRS in each year. For the atmospheric deposition, 14 kg N ha^{-1} $year^{-1}$ were obtained from UBA (2019), representing a mean from 2013 to 2015. The N output included N uptake into aboveground biomass ($N_{biomass}$, calculated as described above) and N leaching ($N_{leaching}$; see Section 2.6), resulting in Equation (1):

$$N_{balance} = N_{fertilizer} + N_{irrigation} + N_{deposition} - N_{biomass} - N_{leaching}. \quad (1)$$

2.9 | Statistical analyses

Statistical analyses were performed with R version 3.5.1 (R Core Team, 2018). The effects of two water regimes, three bioenergy crops and their interaction on the total mineral N content in the 0–90 cm soil horizon, infiltration rates, cumulative N leaching, N uptake through aboveground biomass and the N balance were analysed by a linear mixed-effects model considering the error structure of the split plot design. In all cases, the factor water regime as well as the interaction between crop and water regime were non-significant and therefore removed from the model. Similarly, for all data collected before the first HRS in July 2017, the factor water regime and the interaction between water regime and crop were not considered. Thus, the data shown in subsequent figures include the treatments with and without HRS if not otherwise noted.

The sampling dates for the N_{min} data, the measurement dates for the infiltration data, the measurement periods for N leaching and the years for N uptake into aboveground biomass and the N balance were analysed separately. Residuals of the final model were checked for homoscedasticity by Levene's test as well as graphically and for normal distribution by the Shapiro–Wilk test as well as graphically. The data for the N_{min} contents in March 2017, March 2018 and October 2018, the infiltration rates in each year and the data for N leaching for all periods except summer 2017 were log-transformed, while the data for the N balance in 2017–2018 were root-transformed prior to analysis to fulfil the prerequisites for analysis. If the factor crop was significant ($\alpha \leq 0.05$), means were compared by calculating estimated marginal means (package 'emmeans').

3 | RESULTS

3.1 | Temperature, precipitation and soil moisture

During the study period, the experimental site experienced extremely diverse weather conditions, ranging from above-average precipitation in the 2017 growing season to severe drought combined with high above-average temperatures in the 2018 growing season (Figure 1). Compared to the average long-term March–October rainfall total of 434 mm, abundant rainfall of 618 mm occurred during this period in 2017, but only 223 mm of rain fell in 2018. Because of the exceptionally high precipitation especially during the summer months, soil moisture levels were generally high in 2017, with the highest values occurring in the grass plots and the lowest values in the cup plant plots. This contrasts with the extremely dry year of 2018, when, despite intensive irrigation of the entire experiment, the soil moisture was generally low, particularly in the cup plant and maize plots (Figure 1).

3.2 | Soil mineral N content

The soil mineral N content was generally low, ranging from 9 to 40 kg N_{min}/ha on average in the 0–90 cm soil depth (Figure 2). At all sampling dates except March 2018, cup plant and grass showed significantly lower N_{min} content than maize. At most sampling dates, a larger share of the total N_{min} content was located in the deeper soil horizons of 30–90 cm in maize with 10–24 kg N/ha compared to 4–6 kg N/ha for cup plant and grass.

3.3 | Water infiltration rates, water runoff and soil erosion

The water infiltration rates at the beginning of the vegetation period were significantly different among the crops (Figure 3). In each year, infiltration rates in the cup plant and grass plots were significantly higher than in the maize plots. Infiltration rates under grass were higher than under cup plant in 2017 and 2018 but not in 2019.

While runoff and soil loss were negligible in the cup plant and grass plots throughout all measurement periods, large amounts of runoff water and eroded sediment were collected from the maize plots during the summer of 2017 (Table 1). A high-intensity rainfall with 26 mm/hr in May 2017, 9 days after ploughing of the maize plots, formed a soil crust on the bare maize plots. During the HRS in 2017, more than half of the applied irrigation water was lost to runoff in the maize plots. Soil loss from the maize plots in summer 2017 occurred mostly during the first 6 weeks after the high-intensity rainfall (98% of total soil loss in this period). In the summer of 2018 and the winter 2018–2019, runoff and soil loss were generally low for all crops.

3.4 | N leaching

N leaching in the winter period of 2016–2017 was generally low (Figure 4). While cup plant also did not show high leaching losses during the following summer, significantly higher N leaching for maize and for grass occurred in that period. During the winter of 2017–2018, N leaching in the grass and cup plant plots was low, while maize exhibited significantly higher N leaching losses. Due to dry conditions in the summer of 2018, N leaching was again quite low for all crops. In winter 2018–2019, grass and cup plant showed similarly low N leaching as in the two winters before, while considerable leaching losses for maize were found. In all measurement

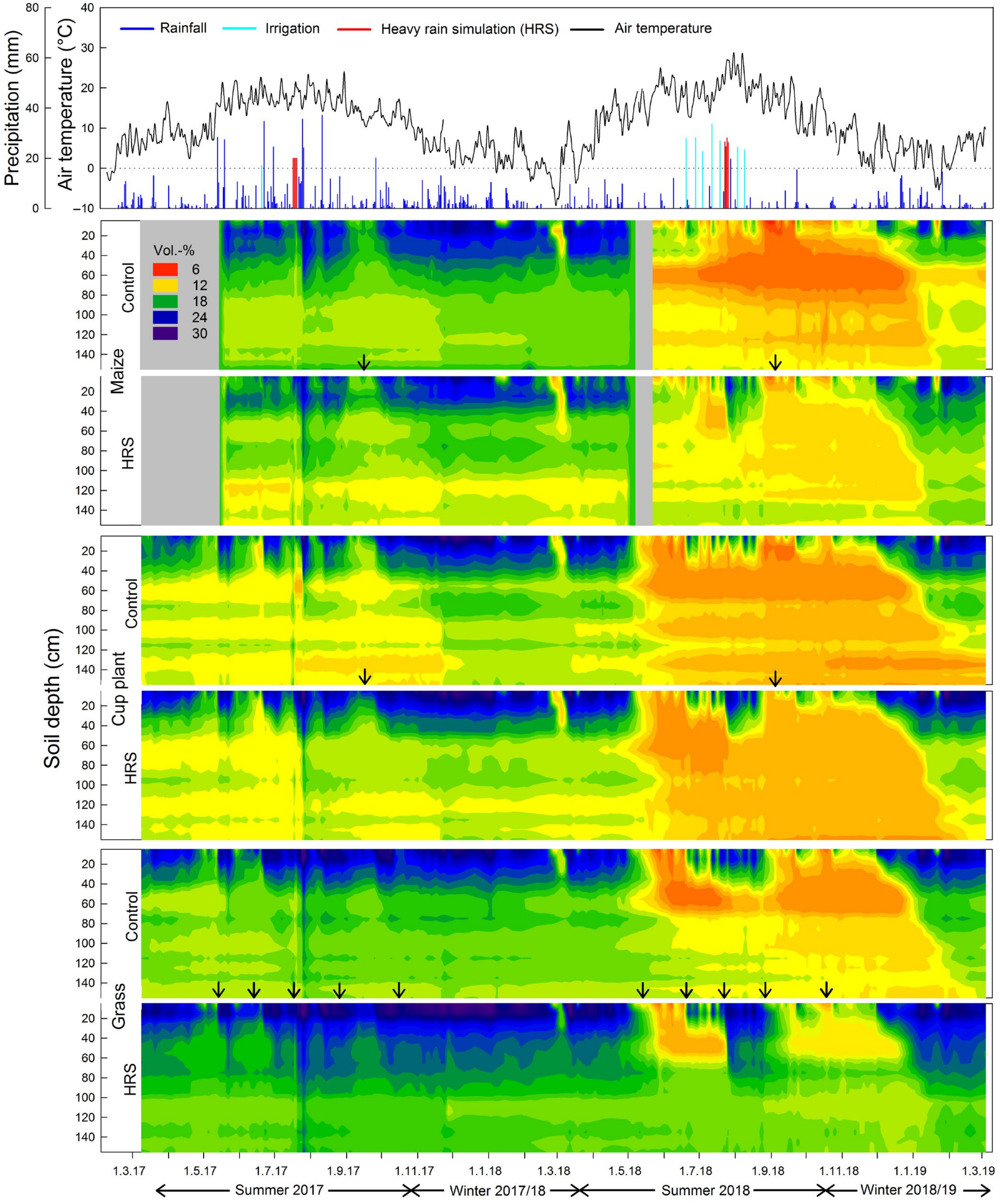

FIGURE 1 Daily precipitation in the form of rainfall, irrigation, and heavy rain simulation (HRS), mean daily 2 m air temperature as well as soil water content (vol.%) in the 0–160 cm soil horizon under maize, cup plant and grass plots and the water regimes with HRS and without (Control) over a 2-year period at Braunschweig, Germany. The HRS was only practised on four of the eight field plots, but irrigation water was provided to all plots when necessary. Vertical arrows denote crop harvest dates which were identical for the two water regimes. Horizontal arrows denote the length of the summer and winter measurement periods for the self-integrating accumulators. No soil moisture measurements were conducted during the 2016–2017 winter period and the periods marked grey in the figure

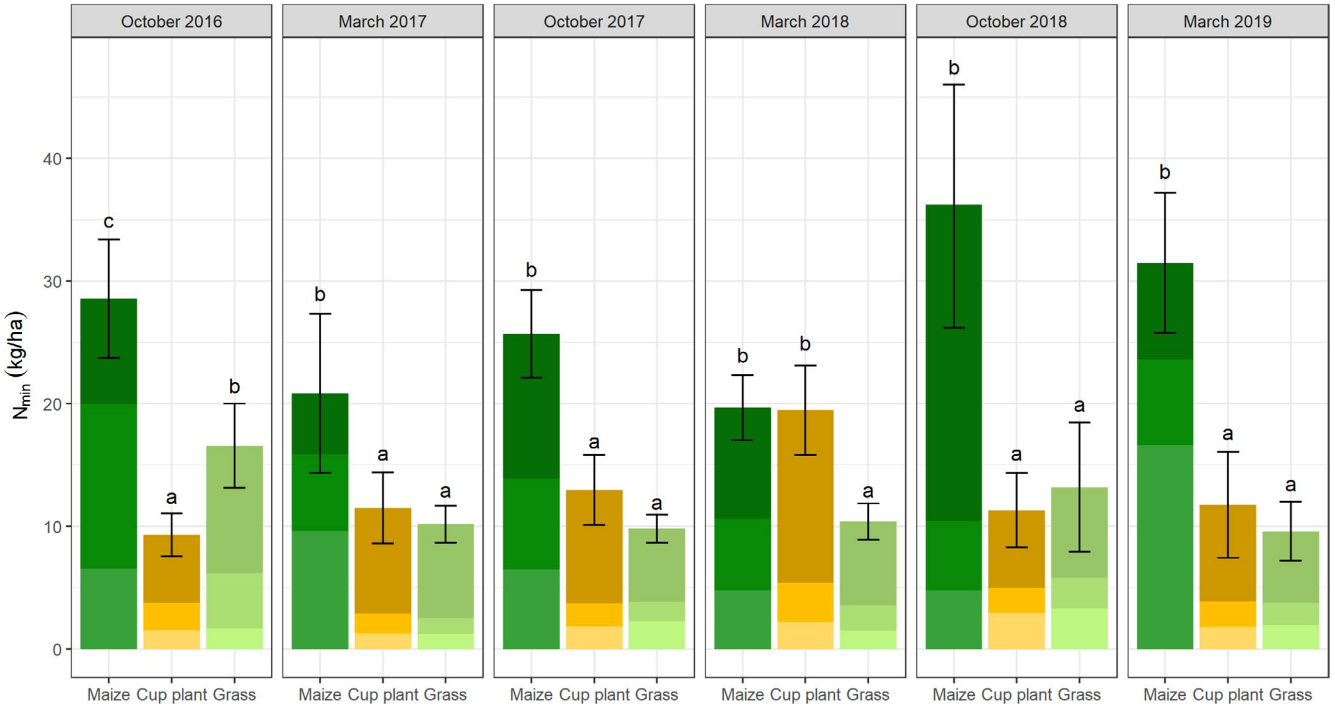

FIGURE 2 Soil mineral N content for the three crops in 0–90 cm soil depth at different sampling dates ($n = 8$). The different shades represent the 0–30, 30–60 and 60–90 cm soil depth from darkest to lightest. Note: The standard deviation is calculated for the whole profile from 0–90 cm soil depth. Different letters indicate significant ($\alpha < 0.05$) differences between crops for the given sampling date. The data for March 2017, March 2018 and October 2018 were log-transformed before analysis

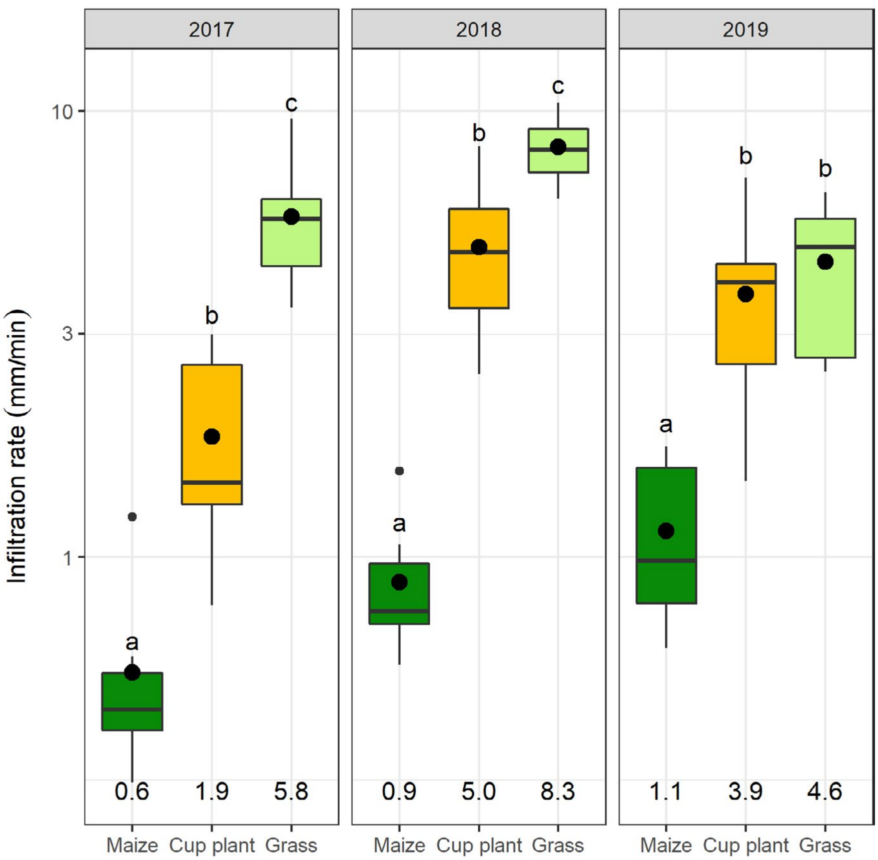

FIGURE 3 Water infiltration rates of the three crops ($n = 8$), measured at the beginning of the vegetation period in three subsequent years. The y-axis is a logarithmic scale. The boxes show the interquartile ranges with the crossbar depicting the median. Large dots and numbers at the bottom show the mean value. Different letters indicate significant ($\alpha < 0.05$) differences between crops for the respective year. Prior to statistical analysis, data for all years had to be log-transformed

TABLE 1 Cumulative water runoff and soil loss for the three crops in different measurement periods and during the heavy rain simulation (HRS) in 2017 and 2018. Data shown are means of four field replicates with standard deviations in brackets

	Maize		Cup plant		Grass	
Leaching period	**Runoff (L/m^2)**	**Soil loss (kg/ha)**	**Runoff (L/m^2)**	**Soil loss (kg/ha)**	**Runoff (L/m^2)**	**Soil loss (kg/ha)**
Winter 2016–2017	1.8 (0.7)	29 (17)	<0.1	0	<0.1	0
Summer 2017	19.4 (2.3)	1,299 (405)	0.2 (0.1)	0	1.0 (1.4)	11 (22)
HRS 2017	46.7 (15.0)	197 (102)	0.0	0	0.0	0
Winter 2017–2018	7.8 (0.1)	0	0.0	0	0.0	0
Summer 2018	0.4 (0.3)	0	0.2 (0.3)	0	<0.1	0
HRS 2018	4.7 (4.5)	45 (39)	<0.1	0	<0.1	0
Winter 2018–2019	1.7 (0.1)	0	0.0	0	0.0	0

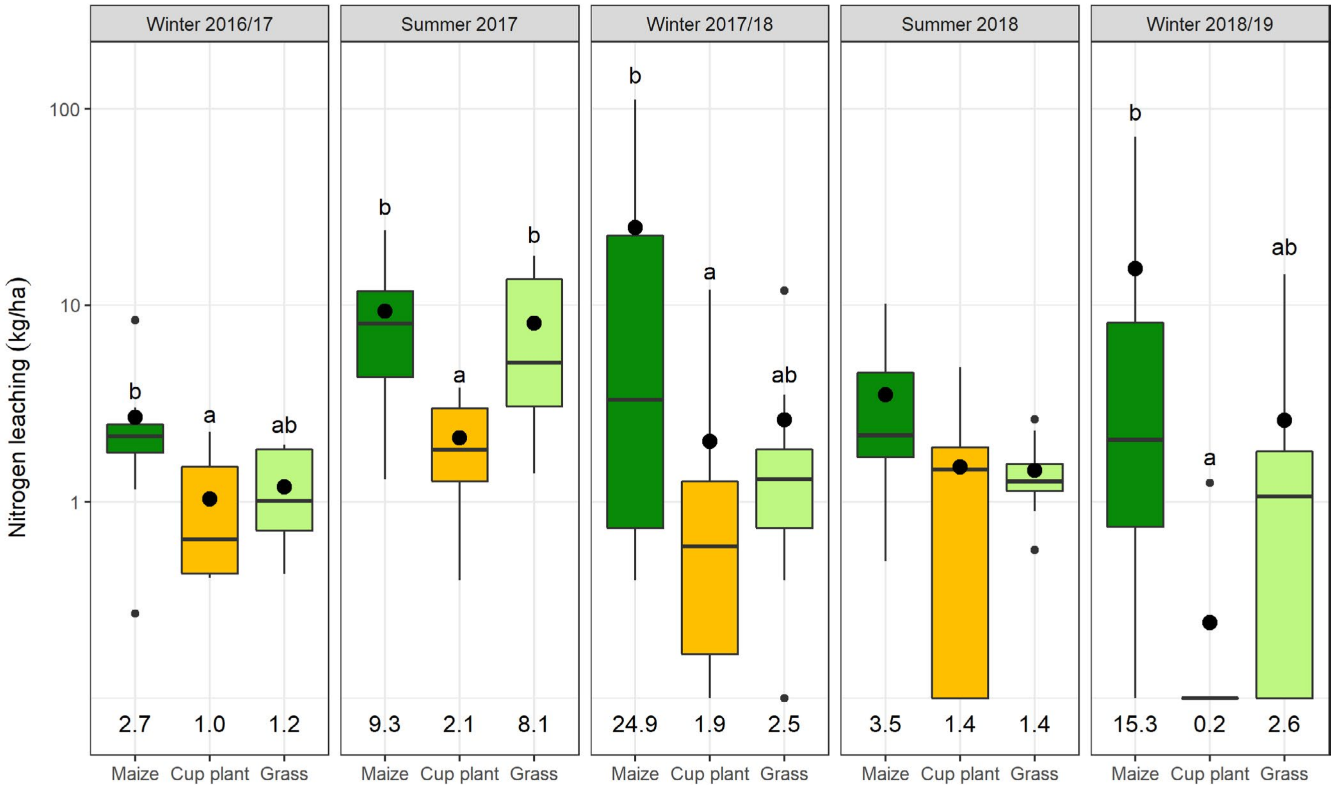

FIGURE 4 Cumulative N leaching (NO_3^--N + NH_4^+-N) in kg N/ha ($n = 8$) below 120 cm soil depth under the three crops in five measurement periods. The y-axis is a logarithmic scale. The boxes show the interquartile ranges with the crossbar depicting the median. Large dots and numbers at the bottom show the mean value. Different letters indicate significant ($\alpha < 0.05$) differences between crops for the respective period. If no letters are shown, no significant differences between crops were found. Prior to statistical analysis, data for all periods, except for summer 2017, had to be log-transformed

periods, except summer 2018, significantly lower N leaching was found for cup plant than for maize.

3.5 | Aboveground biomass, N uptake and N balance

In both years, maize produced the highest dry matter yield averaged across both HRS treatments with 24.7 and 24.5 t/ha, followed by cup plant with 18.2 and 14.9 t/ha and grass with 16.2 and 13.7 t/ha in 2017 and 2018, respectively. N uptake into aboveground biomass generally exceeded the applied amounts of fertilizer-N (Table 2). Additional N uptake into regrown cup plant biomass in November 2019 amounted to 22.2 ± 4.3 kg N/ha (mean and standard deviation of the eight field replicates). The N balances were negative for all crops, except for cup plant in 2018–2019. Maize had the most negative N balance.

TABLE 2 N balance for three crops in the two study years mid-March 2017 to mid-March 2018 (2017–2018) and mid-March 2018 to mid-March 2019 (2018–2019). Data shown are means with standard deviations in brackets of eight field replications. Different letters indicate significant ($\alpha < 0.05$) differences between crops within the respective study year. Data for leaching losses had to be log-transformed and data for the balance in 2017–2018 had to be root-transformed prior to analysis

	Input (kg N/ha)			Output (kg N/ha)		
Study year	**Fertilizer**	**Deposition**	**Irrigation**	**N in biomass**	**Leaching**	**Balance**
2017–2018						
Maize	180	14	7	256 (24)^{b}	34 (47)^{b}	−90 (60)^{a}
Cup plant	170	14	7	207 (26)^{a}	4 (5)^{a}	−20 (24)^{b}
Grass	310	14	7	370 (10)^{c}	11 (10)^{ab}	−50 (15)^{ab}
2018–2019						
Maize	180	14	36	305 (23)^{b}	19 (29)^{b}	−94 (32)^{a}
Cup plant	170	14	36	212 (15)^{a}	2 (2)^{a}	6 (13)^{c}
Grass	310	14	36	376 (17)^{c}	4 (5)^{ab}	−20 (17)^{b}

4 | DISCUSSION

The presented field study started 5 years after the establishment of cup plant and grass. Such a long lead-time is assumed sufficient to detect possible effects of different cropping systems on, for example, earthworm communities as one of the most important factors in mitigating runoff and erosion (Schorpp & Schrader, 2016). Likewise, possible effects on N leaching should also be fully detectable. Hypothesis (1) was confirmed; cup plant and grass had a higher water infiltration than maize and water runoff and soil erosion were reduced to near zero. In contrast, maize showed large amounts of surface runoff and associated soil erosion following natural and artificial heavy rain events. Hypothesis (2) was confirmed; cup plant and grass had lower mineral N content in soil and lost less N via leaching than maize despite higher water infiltration rates. Hypothesis (3) was confirmed; the cultivation of permanent cup plant achieved similar environmental benefits as grass and had an even more favourable N balance. Overall, grass and especially cup plant showed great advantages over maize regarding soil and water protection.

High infiltration rates of the two perennial crops led to near-zero rates of water runoff and soil erosion, while low infiltration rates under maize caused high amounts of runoff and erosion. The higher infiltration in the perennial crops can be related to higher abundances or activity of earthworms in grass (Lamandé et al., 2003; Whalen, 2004) and cup plant (Emmerling, 2014; Emmerling, Schmidt, Ruf, von Francken-Welz, & Thielen, 2017; Schorpp & Schrader, 2016) compared to maize. The increase in earthworm channels and improved soil structure has probably caused the higher infiltration rates (Blouin et al., 2013; Fischer et al., 2014). In perennial switchgrass, higher soil porosity including biopores and the dense root network were also used to explain the observed higher infiltration rates compared to maize (Blanco-Canqui, 2010). In addition, the lower infiltration rates under maize in connection with spring tillage of the maize plots probably caused the formation of the soil crust in the maize plots after a heavy natural rainfall in spring 2017 (Le Bissonnais et al., 2005). The cultivation of maize generally leads to higher levels of runoff and erosion because of the long periods without vegetation cover and the large row distances (Cerdan et al., 2010; Vogel et al., 2016). Given the almost flat experimental field in the present study, the amounts of runoff and sediment erosion from the maize plots in 2017 (1.5 Mg/ha) are still remarkable. Nonetheless, the amount of eroded sediment is well below the European average for arable land (4.4 Mg/ha; Cerdan et al., 2010). Further soil loss through erosion might have been prevented by the surface crust acting as a seal (Valentin & Bresson, 1992) after the first few rainfall events as well as by the developing maize canopy (Ma, Yu, Ma, Li, & Wu, 2014). It should be noted that in this study the classical method of maize cultivation was practiced, that is winter fallow land and ploughing before sowing. Plant cultivation measures such as the use of winter catch crops with subsequent no-till or strip-till sowing of maize are increasingly being practiced in Germany. However, while reduced tillage intensities in maize cultivation might reduce the risk of runoff and erosion (Vogel et al., 2016), N losses through leaching could even increase with this cultivation practices (Daryanto, Wang, & Jacinte, 2017).

In general, croplands with flat slopes similar to the one of the studied site are not prone to erosion. On a steep slope with a much higher erosion potential, Cosentino et al. (2015) found perennial crops to greatly reduce erosion compared to annual crops. Such sites, for example in low mountain ranges, would be particularly suitable growing areas for cup plant, all the more so as the generally higher precipitation in those areas would meet the cup plants' high water demand (Schoo, Wittich, et al., 2017). The observed prevention of runoff and erosion by cup plant might be enhanced in a changing climate, with meteorological extremes such as heavy rainfall generally becoming more likely (O'Gorman, 2015). Even narrow stripes of perennial crops below 10 m width placed in the preferred drainage channels of steep hills might suffice to significantly reduce runoff and soil erosion (Kreig, Ssegane, Chaubey, Negri, & Jager, 2019).

Lower N_{min} contents under cup plant and grass, particularly below 30 cm soil depth, compared to maize led to lower N leaching in all seasons. The lower N_{min} levels under both perennial crops in comparison with maize might be attributed to the longer phase of N uptake caused by an earlier plant growth and the regrowth after harvest (Pugesgaard et al., 2015). In fact, the N uptake (22.2 kg N/ha) by regrowth of cup plant after the harvest in 2019 corresponds well to the order of magnitude of differences in N_{min} contents (13–29 kg N/ha) under cup plant and maize at the three October sampling dates. The N_{min} contents under maize were particularly high in the 30–90 cm soil horizon, which might explain the higher N leaching losses from the maize plots in the wet period from mid-March 2017 to mid-March 2018. The annual amount of leached N from maize in the above-mentioned period was comparable to the global average leaching loss for maize (30.6 kg N/ha; Zhou & Butterbach-Bahl, 2014) with comparable amounts of N fertilization. Furthermore, similar mean N leaching losses of 34.3 kg N/ha following maize cultivation were obtained with SIA measurements at many locations and over several years in Germany and Switzerland (TerrAquat Consultants, unpublished data). Lower N leaching for perennial compared to annual crops was found previously, for various combinations of crops (Ferchaud & Mary, 2016; Hussain et al., 2019; McIsaac et al., 2010; Pugesgaard et al., 2015). Besides lower N_{min} contents, the lower N leaching could also be partly attributable to lower seepage (Pugesgaard et al., 2015). In the case of cup plant, the well-known high water demand (Schoo, Wittich, et al., 2017) strongly limits the downward water transport. Thus, the growing area of cup plant might also extend to sites with direct connection to the groundwater and thus a high risk of N leaching.

The benefits of cup plant with regard to soil and water protection were similar to those of grass. Moreover, while also showing negligible N losses to the environment in both years, cup plant had a near-zero N balance. Grass and especially maize had a negative N balance indicating a loss of soil organic matter by mineralization. Fernández, Fabrizzi, and Naeve (2017) found annual rates of N mineralization under maize between 86 and 134 kg N/ha, which corresponds well to the difference between N uptake and N fertilization for the maize plots in the present field experiment. The loss of organic N under maize might have been even greater than shown in our data, as the potential N_2O emission was not examined. Higher N_2O emissions have been found for annual compared to perennial crops (Abalos et al., 2016; Smith et al., 2013) and would thus add to the N deficit of maize compared to cup plant and grass. The suspected higher mineralization in the maize plots can be attributed partly to the annual tillage operations (Kaiser et al., 2014; Van den Bossche, de Bolle, de Neve, & Hofman, 2009). Thus, despite showing higher N losses to the environment by leaching, maize cropping also depletes the soil organic N pools when no cultivation measures are taken to counterbalance these losses.

5 | CONCLUSIONS

In this study, cup plant and grass had similar positive environmental effects, with slight advantages for the cup plant. The benefits of cup plant in terms of soil and water protection may be even greater on sites with sloping terrain and high groundwater levels. It should therefore be examined to what extent the results obtained in this study can be transferred to such particularly sensitive environments. Overall, it would be desirable to increase the cup plant acreage to an ecologically relevant scale. An important prerequisite was a recent breakthrough in the cultivation of cup plant (Fröhlich, Brodmann, & Metzler, 2016). By sowing seeds instead of planting seedlings, establishment costs could be reduced by about 60% (TLL, 2018). In addition, by under-sowing in maize, the first year of cup plant cultivation can be combined with a maize biomass harvest. So far, cup plant has only produced an economically unattractive biomass yield in the year of establishment. Partly as a result of this innovation, the cup plant area in Germany has increased from a few hundred hectares in 2015 to about 4,500 ha in 2019.

ACKNOWLEDGEMENTS

The authors thank Friederike Diers, Sabine Peickert, Martina Schabanoski, Bernd Kahlstorf, Birgit Pohl, Martina Küchenthal, Andrea Schmidt-Meier, Fredrik Mühlberger and Timo Schmidt for their help in executing the field experiments and laboratory analyses. The authors also thank David Fernall for proofreading the manuscript.

AUTHORS' CONTRIBUTIONS

S.S., K.P. and W.A.B. designed the experiment. D.G. performed the statistical analyses. D.G. and S.S. wrote the manuscript. Each author participated in the critical discussion of the results of the study during the composing of the manuscript and revised the manuscript.

ORCID

Dennis Grunwald https://orcid.org/0000-0002-2621-2971
Kerstin Panten https://orcid.org/0000-0003-3723-7549
Siegfried Schittenhelm https://orcid.org/0000-0002-2743-0989

REFERENCES

Abalos, D., Brown, S. E., Vanderzaag, A. C., Gordon, R. J., Dunfield, K. E., & Wagner-Riddle, C. (2016). Micrometeorological measurements over 3 years reveal differences in N_2O emissions between annual and perennial crops. *Global Change Biology*, *22*, 1244–1255. https://doi.org/10.1111/gcb.13137

Angulo-Jaramillo, R., Bagarello, V., Iovino, M., & Lassabatere, L. (2016). *Infiltration measurements for soil hydraulic characterization*. Switzerland: Springer International Publishing AG.

Bischoff, W. A. (2007). *Development and applications of the self-integrating accumulators: A method to quantify the leaching losses of environmentally relevant substances*. Germany: Technical University of Berlin. Retrieved from https://doi.org/10.14279/depositonce-1766

Blanco-Canqui, H. (2010). Energy crops and their implications on soil and environment. *Agronomy Journal*, *102*, 403–419. https://doi.org/10.2134/agronj2009.0333

Blouin, M., Hodson, M. E., Delgado, E. A., Baker, G., Brussaard, L., Butt, K. R., … Brun, J. J. (2013). A review of earthworm impact on soil function and ecosystem services. *European Journal of Soil Science*, *64*, 161–182. https://doi.org/10.1111/ejss.12025

BMU (Federal Ministry of the Environment, Nature Conservation and Nuclear Safety) & BMELV (Federal Ministry of Food, Agriculture and Consumer Protection). (2016). *Nitrate report 2016*. Bonn, Germany: BMU (Federal Ministry of the Environment, Nature Conservation and Nuclear Safety) & BMELV (Federal Ministry of Food, Agriculture and Consumer Protection).

Cerdan, O., Govers, G., Le Bissonnais, Y., Van Oost, K., Poesen, J., Saby, N., … Dostal, T. (2010). Rates and spatial variations of soil erosion in Europe: A study based on erosion plot data. *Geomorphology*, *122*, 167–177. https://doi.org/10.1016/j.geomorph.2010.06.011

Cosentino, S. L., Copani, V., Scalici, G., Scordia, D., & Testa, G. (2015). Soil erosion mitigation by perennial species under Mediterranean environment. *Bioenergy Research*, *8*, 1538–1547. https://doi.org/10.1007/s12155-015-9690-2

Daryanto, S., Wang, L., & Jacinte, P.-A. (2017). Impacts of no-tillage management on nitrate loss from corn, soybean and wheat cultivation: A meta-analysis. *Scientific Reports*, *7*(1), 12117. https://doi.org/10.1038/s41598-017-12383-7

de Ruijter, F. J., Boumans, L. J. M., Smit, A. L., & van den Berge, M. (2007). Nitrate in upper groundwater on farms under tillage as affected by fertilizer use, soil type and groundwater table. *Nutrient Cycling in Agroecosystems*, *77*, 155–167. https://doi.org/10.1007/s10705-006-9051-9

Emmerling, C. (2014). Impact of land-use change towards perennial energy crops on earthworm population. *Applied Soil Ecology*, *84*, 12–15. https://doi.org/10.1016/j.apsoil.2014.06.006

Emmerling, C., Schmidt, A., Ruf, T., von Francken-Welz, H., & Thielen, S. (2017). Impact of newly introduced perennial bioenergy crops on soil quality parameters at three different locations in W-Germany. *Journal of Plant Nutrition and Soil Science*, *180*, 759–767. https://doi.org/10.1002/jpln.201700162

EUR-Lex. (2018). *Commission delegated regulation (EU) 2018/1784*. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1569312063105&uri=CELEX:32018R1784

Ferchaud, F., & Mary, B. (2016). Drainage and nitrate leaching assessed during 7 years under perennial and annual bioenergy crops. *Bioenergy Research*, *9*, 656–670. https://doi.org/10.1007/s12155-015-9710-2

Fernández, F. G., Fabrizzi, K. P., & Naeve, S. L. (2017). Corn and soybean's season-long in-situ nitrogen mineralization in drained and undrained soils. *Nutrient Cycling in Agroecosystems*, *107*, 33–47. https://doi.org/10.1007/s10705-016-9810-1

Fiener, P., Neuhaus, P., & Botschek, J. (2013). Long-term trends in rainfall erosivity – Analysis of high resolution precipitation time series (1937–2007) from Western Germany. *Agricultural and Forest Meteorology*, *171–172*, 115–123. https://doi.org/10.1016/j.agrformet.2012.11.011

Fischer, C., Roscher, C., Jensen, B., Eisenhauer, N., Baade, J., Attinger, S., … Hildebrandt, A. (2014). How do earthworms, soil texture, and plant composition affect infiltration along an experimental plant diversity gradient in grassland? *PLoS One*, *9*(6), e98987. https://doi.org/10.1371/journal.pone.0098987

FNR. (2018). *Basisdaten Bioenergie Deutschland 2018*. Retrieved from http://www.fnr.de/fileadmin/allgemein/pdf/broschueren/Basisdaten_Bioenergie_2018.pdf

Fröhlich, W., Brodmann, R., & Metzler, T. (2016). The cup plant (*Silphium perfoliatum* L.) – A story of success from agricultural practice. *Journal of Cultivated Plants*, *68*, 351–355. https://doi.org/10.1399/JFK.2016.12.01 (in German).

Gauder, M., Billen, N., Zikeli, S., Laub, M., Graeff-Hönninger, S., & Claupein, W. (2016). Soil carbon stocks in different bioenergy cropping systems including subsoil. *Soil and Tillage Research*, *155*, 308–317. https://doi.org/10.1016/j.still.2015.09.005

Haag, N. L., Nägele, H. J., Reiss, K., Biertümpfel, A., & Oechnser, H. (2015). Methane formation potential of cup plant (*Silphium perfoliatum*). *Biomass and Bioenergy*, *75*, 126–133. https://doi.org/10.1016/j.biombioe.2015.02.012

Hussain, M. Z., Bhardwaj, A. K., Basso, B., Robertson, G. P., & Hamilton, S. K. (2019). Nitrate leaching from continuous corn, perennial grasses, and poplar in the US Midwest. *Journal of Environmental Quality*, *48*, 1849–1855. https://doi.org/10.2134/jeq2019.04.0156

IUSS Working Group WRB. (2015). *World Reference Base for Soil Resources 2014, update 2015. International soil classification system for naming soils and creating legends for soil maps*. World Soil Resources reports no. 106. Rome, Italy: FAO.

Kaiser, M., Piegholdt, C., Andruschkewitsch, R., Linsler, D., Koch, H.-J., & Ludwig, B. (2014). Impact of tillage intensity on carbon and nitrogen pools in surface and sub-surface soils of three long-term field experiments. *European Journal of Soil Science*, *65*, 499–509. https://doi.org/10.1111/ejss.12146

Köhler, K., Duynisveld, W. H. M., & Böttcher, J. (2006). Nitrogen fertilization and nitrate leaching into ground water on arable sandy soils. *Journal of Plant Nutrition and Soil Science*, *169*, 185–195. https://doi.org/10.1002/jpln.200521765

Kreig, J. A. F., Ssegane, H., Chaubey, I., Negri, M. C., & Jager, H. I. (2019). Designing bioenergy landscapes to protect water quality. *Biomass and Bioenergy*, *128*, 105327. https://doi.org/10.1016/j.biombioe.2019.105327

Lamandé, M., Hallaire, V., Curmi, P., Pérès, G., & Cluzeau, D. (2003). Changes of pore morphology, infiltration and earthworm community in a loamy soil under different agricultural managements. *Catena*, *54*, 637–649. https://doi.org/10.1016/S0341-8162(03)00114-0

Le Bissonnais, Y., Cerdan, O., Lecomte, V., Benkhadra, H., Souchère, V., & Martin, P. (2005). Variability of soil surface characteristics influencing runoff and interrill erosion. *Catena*, *62*, 111–124. https://doi.org/10.1016/j.catena.2005.05.001

Ma, B., Yu, X., Ma, F., Li, Z., & Wu, F. (2014). Effects of crop canopies on rain splash detachment. *PLoS One*, *9*(7), e99717. https://doi.org/10.1371/journal.pone.0099717

Mast, B., Lemmer, A., Oechsner, H., Reinhardt-Hanisch, A., Claupein, W., & Graeff-Hönninger, S. (2014). Methane yield potential of novel perennial biogas crops influenced by harvest date. *Industrial Crops and Products*, *58*, 194–203. https://doi.org/10.1016/j.indcrop.2014.04.017

McIsaac, G. F., David, M. B., & Mitchell, C. A. (2010). Miscanthus and switchgrass production in Central Illinois: Impacts on hydrology and inorganic nitrogen leaching. *Journal of Environmental Quality*, *39*, 1790–1799. https://doi.org/10.2134/jeq2009.0497

O'Gorman, P. A. (2015). Precipitation extremes under climate change. *Current Climate Change Reports*, *1*, 49–59. https://doi.org/10.1007/s40641-015-0009-3

Palmer, R. C., & Smith, R. P. (2013). Soil structural degradation in SW England and its impact on surface-water runoff generation. *Soil Use and Management*, *29*, 567–575. https://doi.org/10.1111/sum.12068

Pugesgaard, S., Schelde, K., Larsen, S. U., Lærke, P. E., & Jørgensen, U. (2015). Comparing annual and perennial crops for bioenergy production – Influence on nitrate leaching and energy balance. *GCB Bioenergy*, *7*, 1136–1149. https://doi.org/10.1111/gcbb.12215

R Core Team. (2018). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.

Routschek, A., Schmidt, J., Enke, W., & Deutschlaender, T. (2014). Future soil erosion risk – Results of GIS-based model simulations for a catchment in Saxony/Germany. *Geomorphology*, *206*, 299–306. https://doi.org/10.1016/j.geomorph.2013.09.033

Schoo, B., Schroetter, S., Kage, H., & Schittenhelm, S. (2017). Root traits of cup plant, maize and lucerne grass grown under different soil and soil moisture conditions. *Journal of Agronomy and Crop Science*, *203*, 345–359. https://doi.org/10.1111/jac.12194

Schoo, B., Wittich, K. P., Böttcher, U., Kage, H., & Schittenhelm, S. (2017). Drought tolerance and water-use efficiency of biogas crops: A comparison of cup plant, maize and lucerne-grass. *Journal of Agronomy and Crop Science*, *203*, 117–130. https://doi.org/10.1111/jac.12173

Schorpp, Q., & Schrader, S. (2016). Earthworm functional groups respond to the perennial energy cropping system of the cup plant (*Silphium perfoliatum* L.). *Biomass and Bioenergy*, *87*, 61–68. https://doi.org/10.1016/j.biombioe.2016.02.009

Schwärzel, K., & Punzel, J. (2007). Hood infiltrometer – A new type of tension infiltrometer. *Soil Science Society of America Journal*, *71*, 1438–1447. https://doi.org/10.2136/sssaj2006.0104

Smith, C. M., David, M. B., Mitchell, C. A., Masters, M. D., Anderson-Teixeira, K. J., Bernacchi, C. J., & DeLucia, E. H. (2013). Reduced nitrogen losses after conversion of row crop agriculture to perennial biofuel crops. *Journal of Environmental Quality*, *42*, 219–228. https://doi.org/10.2134/jeq2012.0210

TLL. (2018). *Leitlinie zur effizienten und umweltverträglichen Erzeugung von Durchwachsener Silphie*. Retrieved from http://www.tll.de/www/daten/publikationen/leitlinien/ll_silphie.pdf

UBA. (2019). *Hintergrundbelastungsdaten Stickstoff*. Umweltbundesamt. Retrieved from https://gis.uba.de/website/depo1/

Valentin, C., & Bresson, L.-M. (1992). Morphology, genesis and classification of surface crusts in loamy and sandy soils. *Geoderma*, *55*, 225–245. https://doi.org/10.1016/0016-7061(92)90085-L

Van den Bossche, A., De Bolle, S., De Neve, S., & Hofman, G. (2009). Effect of tillage intensity on N mineralization of different crop residues in a temperate climate. *Soil and Tillage Research*, *103*(2), 316–324. https://doi.org/10.1016/j.still.2008.10.019

Vogel, E., Deumlich, D., & Kaupenjohann, M. (2016). Bioenergy maize and soil erosion – Risk assessment and erosion control concepts. *Geoderma*, *261*, 80–92. https://doi.org/10.1016/j.geoderma.2015.06.020

Weiland, P. (2010). Biogas production: Current state and perspectives. *Applied Microbiology and Biotechnology*, *85*, 849–860. https://doi.org/10.1007/s00253-009-2246-7

Whalen, J. (2004). Spatial and temporal distribution of earthworm patches in corn field, hayfield and forest systems of southwestern Quebec, Canada. *Applied Soil Ecology*, *27*, 143–151. https://doi.org/10.1016/j.apsoil.2004.04.004

Zhou, M., & Butterbach-Bahl, K. (2014). Assessment of nitrate leaching loss on a yield-scaled basis from maize and wheat cropping systems. *Plant and Soil*, *374*, 977–991. https://doi.org/10.1007/s11104-013-1876-9

Zolina, O., Simmer, C., Kapala, A., Bachner, S., Gulev, S., & Maechel, H. (2008). Seasonally dependent changes in precipitation extremes over Germany since 1950 from a very dense observational network. *Journal of Geophysical Research*, *113*, D06110. https://doi.org/10.1029/2007JD008393

How to cite this article: Grunwald D, Panten K, Schwarz A, Bischoff W-A, Schittenhelm S. Comparison of maize, permanent cup plant and a perennial grass mixture with regard to soil and water protection. *GCB Bioenergy*. 2020;12:694–705. https://doi.org/10.1111/gcbb.12719