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RESEARCH ARTICLE

Greenhouse gas emissions from *Silphium perfoliatum* and silage maize cropping on Stagnosols

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Abstract

Background: The sustainability of bioenergy is strongly affected by direct field-derived greenhouse gas (GHG) emissions and indirect emissions form land-use change. Marginal land in low mountain ranges is suitable for feedstock production due to small impact on indirect land-use change. However, these sites are vulnerable to high N_2O emissions because of their fine soil texture and hydrology.

Aims: The perennial cup plant (*Silphium perfoliatum* L.) might outperform silage maize (*Zea mays* L.) on cold, wet low mountain ranges sites regarding yield and ecosystem services. The aim of this study was to assess whether the cultivation of cup plant also provides GHG mitigation potential compared to the cultivation of maize.

Methods: A t-year field experiment was conducted in a low mountain range region in western Germany to compare area and yield-scaled GHG emissions from cup plant and maize fields. GHG emissions were quantified using the closed chamber method.

Results: Cup plant fields emitted an average of 3.6 ± 4.3 kg N_2O-N ha^{-1} $year^{-1}$ (−85%) less than maize fields. This corresponded to 74.0 ± 94.1 g $CO_{2\text{-eq}}$ kWh^{-1} (−78%) less emissions per produced electrical power. However, cup plant had a significantly lower productivity per hectare (−34%) and per unit of applied nitrogen (−32%) than maize.

Conclusion: Cup plant as a feedstock reduces direct field-derived GHG emissions compared to maize but, due to lower yields cup plant, likely increases emissions associated with land-use changes. Therefore, the increased sustainability of bioenergy from biogas by replacing maize with cup plant is heavily dependent on the performance of maize at these sites and on the ecosystem services of cup plant in addition to GHG savings.

KEYWORDS
bioenergy, biomass, cup plant, greenhouse gas, nitrogen, N_2O emission

1 | INTRODUCTION

First-generation biogas production is currently a wide-spread practice in Europe, particularly in Germany, where it uses around 14% of Germany's cropland (FNR, 2021). Energy crops compete directly with food production for arable land (Haberl et al., 2012), and the greenhouse gas (GHG) mitigation potential from substituting fossil energy with electricity produced from biogas is relatively low compared with other feedstocks, such as waste, when GHG emissions from land-use change are taken into account (Giuntoli et al., 2015; Shukla et al., 2019). Nonetheless, bioenergy from biogas is a valuable energy source because it is capable of providing base load power. By processing the biogas into biomethane and bio-liquefied natural gas (Bio-LNG), it can partially replace natural gas and fossil fuels. Therefore, the European renewable energy directive (EU RED II) aims to restrict biogas production from biomass in regions suitable for food production, in order to avoid competition for high-quality arable land and emissions due to indirect land-use change (EC, 2018).

In central Europe, low mountain range regions with their lower crop productivity (BGR, 2021) offer production areas where biomass production has a relatively low opportunity cost (Jiang et al., 2021). However, the main biomass crop for biogas production in central Europe, and Germany in particular, is maize (*Zea mays* L.), which is not necessarily suitable for hillsides due to the high soil erosion potential. Additionally, the expansion of maize production in the last two decades, linked to subsidized biogas production (DESTATIS, 2022; DMK, 2022), has led to concerns about reduced landscape diversity due to the omnipresence of maize, loss of soil organic matter (SOM) and biodiversity, and soil compaction (Capriel, 2013; Emmerling, 2014; Linhart & Dhungel, 2013; Ruf & Emmerling, 2018; Schorpp & Schrader, 2016).

The perennial herbaceous biomass crop cup plant (*Silphium perfoliatum* L.) is a promising alternative to maize since it provides a broad palette of ecosystem services, ranging from soil preservation to greater attractiveness to pollinators (Bufe and Korevaar, 2018; Gansberger et al., 2015), and thus adds more diversity to rural areas than most common annual crops. However, cup plant does not achieve as high biomass and methane yields as maize under favorable conditions (von Cossel et al., 2020). The opportunity costs for GHG mitigation on productive sites are high if a low-yielding crop replaces a high-yielding crop (Searchinger et al., 2009). In a greenhouse experiment, Ruf and Emmerling (2018) demonstrated that higher biogas yields are possible from cup plant than from maize if the crops are cultivated on slow-draining, fine-textured soils less suitable for maize production. The simulated soil conditions in their experiment were typical of low mountain ranges. Grunwald et al. (2020) also suggested that low mountain ranges are particularly suitable for cup plant cultivation because of the benefits for soil protection and their ability to meet the crop's high water requirement.

Soil-derived GHG emissions from crop production are an important aspect when evaluating an alternative crop for feedstock production (EC, 2018; Jin et al., 2019). Gaseous N losses as nitrous oxide (N_2O) are a particular concern because it is a potent GHG and resulting in ozone depletion in the stratosphere (Ravishankara et al., 2009). N_2O, an intermediate and by-product of the microbial N cycle in soils, has a global warming potential 265 times greater that of CO_2 over a 100-year time horizon (GWP100; Myhre et al., 2013). The two major processes responsible for N_2O formation in agricultural soils are nitrification and, in particular, denitrification, which causes most of the N_2O emissions from agricultural soils (Butterbach-Bahl et al., 2013; Li et al., 2016).

Denitrification in soils is controlled by soil aeration, readily available carbon (C) sources, and available nitrate (NO_3^-) (Firestone, 1982). These drivers of denitrification are affected by land use and cropping intensity (Aulakh et al., 1992; Ruser et al., 2001). Land use and management practices have an effect on C and N turnover and soil structure, and thus affect soil gas diffusivity (Dobbie & Smith, 2001; Palm et al., 2014). Management practices differ between annual crop rotations and the cultivation of a perennial crop, for example, the frequency of tillage operations and the use of cover crops. Hence, it can be assumed that drivers of denitrification vary substantially between annual maize and perennial cup plant cropping systems.

Few studies have compared perennial biomass crops with annual crops with regard to management-related and induced N_2O emissions. Similar to conventionally managed systems, maize involves frequent tillage operations, while, similar to a no-till system, there is no tillage in cup plant production after establishment. On sites with a higher clay content, higher N_2O emissions have been observed from no-till systems than from conventionally tilled soil, presumably because soil aeration is reduced in the absence of frequent soil loosening (Ball et al., 1999; Rochette, Angers, Chantigny, & Bertrand, 2008; Rochette, Angers, Chantigny, Gagnon et al., 2008). Subsoil compaction due to lithogenesis or pedogenesis (e.g., solifluction) and high clay contents result in slow drainage and cause temporal waterlogging, which is typical for soils in the low mountain ranges of central Europe (Blume et al., 2016). Such soil moisture conditions result in strongly reduced soil gas diffusivity. Therefore, particularly in no-till systems, the combination of soil moisture and soil structure could favor N_2O reduction to N_2 due to the prolonged residence time of N_2O in the soil, which can mitigate N_2O emissions (Chapuis-Lardy et al., 2007). Since the reduction of N_2O to N_2 is strongly linked to the availability of labile C as an energy soucre (Morley & Baggs, 2010), this might be particularly relevant in perennial cropping systems, which are associated with increased accumulation of soil organic carbon (SOC) (Gauder et al., 2016; Lemus & Lal, 2005). In contrast, frequent soil disturbance in annual systems ensures better aeration and lowers the risk of denitrification in poorly aerated soil (Palm et al., 2014). However, frequent tillage can increase mineralization of physically protected SOM, which accelerates oxygen (O_2) consumption (Six et al., 1998). In combination with high soil moisture, the exposure of SOM and the incorporation of organic material, for example, organic fertilizer, and cover crops, by tillage can also promote denitrification in annual cropping systems through increased C availability and respiration (Baggs et al., 2000; Balesdent et al., 2000). Hence, the potential N_2O source strength of perennial no-till and annually tilled soils is not clear and both maize and cup plant have the potential to enhance denitrification through C availability and O_2 limitations.

Maize commonly receives high fertilizer rates before seeding that are applied when the soil temperature is >8°C (Wilson et al., 1995), and

therefore coincides with higher soil microbial activity, which increases the risk of denitrification. In contrast, cup plant commonly receives fertilizer at the beginning of the growing season when soil temperatures are low, and the crop starts taking up the available N immediately due to thorough rooting. Juvenile maize is not capable of depleting mineral nitrogen (N_{min}) rapidly because resource acquisition is limited in the early stages of the crop. Moreover, in dry years, fertilizer that is applied late can remain unused in the soil and is at risk for denitrification losses and leaching in the fall and winter. Thus, maize cropping presents a greater risk of high N_{min} contents and N losses. For cup plant and other perennial crops, longer phases of N uptake diminish the available N_{min} pool over a longer period of time (Grunwald et al., 2020; Pugesgaard et al., 2015).

Annual cropping systems are more prone to high management-related N_2O emission occurring in hot moments than a perennial system, as shown by perennial biomass crops exhibiting lower N_2O emissions than their annual counterparts (Don et al., 2012). However, studies addressing soil-derived GHG emissions from annual and/or perennial bioenergy crops have often compared low-input perennial crops, that is, switchgrass (*Panicum virgatum* L.), miscanthus (*Miscanthus* × *giganteus*), and short rotation coppice, with high-input annual crops, that is, maize, oilseed rape (*Brassica napus* L.), wheat (*Triticum aestivum* L.), triticale (× Triticosecale), and sorghum (*Sorghum bicolor*) (Drewer et al., 2012; Ferchaud et al., 2020; Gauder et al., 2012; Wile et al., 2014). According to Ruf and Emmerling (2021), with its higher nutrient requirements cup plant should not be considered a low-input perennial biomass crop in terms of nutrient substitution.

To evaluate cup plant as a suitable alternative to maize for biogas production on marginally productive sites for C_4 crops with high GHG emission potential, yield-scaled emissions and productivity of the perennial and the annual cropping system have to be considered. Therefore, the objectives of this study were to compare field N_2O and CH_4 emissions from cup plant and silage maize cropping systems, and evaluate the GHG mitigation potential of cup plant cropping based on yield-scaled emissions. We hypothesized that: (1) cultivation of cup plant reduces soil-derived N_2O emissions in biomass production compared with silage maize on marginal sites that are prone to temporal waterlogging; (2) applied nitrogen in cup plant cultivation results in reduced N_2O emissions than in maize cropping due to lower N_{min} availability owing to a longer lasting phase of N uptake in the perennial crop; and (3) biomass and electrical power yield-scaled N_2O emissions from cup plant cropping are comparable to or lower than those from silage maize, although cup plant produces lower yields.

2 | MATERIALS AND METHODS

2.1 | Study sites and experimental design

A field experiment was conducted from February 2019 to February 2021 in the Saar-Nahe low mountain range in the Federal State of Saarland, Germany. The soil conditions were characterized by stagnating soil water resulting in episodic waterlogging, typically from late autumn to early spring. These properties are caused by dense, virtually impermeable soil layers of silty clay or clay loam texture that remain from in situ weathering of the sedimentary rock material from the Cisuralian and upper Pennsylvanian periods (Cohen et al., 2013; Konzen &Müller, 1989). The impermeable soil layer, a Bg-horizon according to the World Reference Base (WRB) (WRB, 2015), is usually present, starting 50 cm below the surface, and is around 20- to 30-cm thick. The overlaying Eg and Ah soil horizons originate from loess material deposited in the last glacial epoch that has been relocated by solifluction. The topsoil horizons are almost free of stones and have a silt-loam texture. Due to the shallowness of the soils, their yield potential is low. According to the WRB (2015), these soils are classified as Hypereutric Mollic Stagnosol (Aric, Loamic) or, if the Bg layer starts below 50 cm, as Hypereutric, Stagnic Cambisol (Aric, Humic, Loamic). Due to the slope inclination of these sites and the texture, they are prone to soil erosion.

The 30-year annual average precipitation and temperature at the nearby weather station (DWD, 2021; station ID 5029) are 1031 mm and 9°C, respectively. The distance between the weather station and the sites ranged from 4.5 to 15.0 km.

The experiment was conducted at four sites (Gronig, Remmesweiler, Dörrenbach, and Fürth; Table 1) consisting of one cup plant (*Silphium perfoliatum*) and one maize (*Zea mays*) field per site. Field sizes ranged between 1 and 5 ha. All fields were managed by the farmers in accordance with common local practice (see Supporting Information for management details). Maize and cup plant fields received on average 200 kg N ha^{-1} $year^{-1}$. The organic fertilizer at all sites was broadcasted, and was only incorporated with a cultivator in maize fields. All cup plant stands were established in 2017. At the Gronig and Remmesweiler sites, maize followed a cereal crop, while in Dörrenbach and Fürth maize had been grown continuously. All the maize fields were cultivated with conservation tillage and a winter cover crop preceded maize in the crop rotation. However, weather conditions at the Remmesweiler site in the fall of 2019 did not allow cover crop establishment. Cup plant fields did not receive any tillage operations after establishment.

Trace gas measurements were conducted on each cup plant and maize field per site (n = 4), with the crop site combination referred to below as a site-pair. Each field was equipped with a set of four collars and chambers (n = 4), resulting in replicated measurements within fields and summing up in a total of 16 chambers in each cropping system. Within the fields, the chambers were installed in measuring plots that were established at locations that were selected based on soil type and properties and position on the slope in order to enable comparability between the fields at each site-pair.

2.2 | Gas flux measurements

Trace gas fluxes were determined weekly, and twice a week after fertilization, using the closed chamber technique (Hutchinson and Mosier,

TABLE 1 Site description and soil texture classification, pH, soil organic carbon (SOC), and total nitrogen (N_t) of the ap horizon (0–30 cm) of each field ($n = 4$ for cup plant and maize)

Site	Crop	Coordinates	Aspect	Altitude (m asl)	Slope (°)	Soil texture	pH	SOC (g C kg^{-1})	N_t (g N kg^{-1})
Gronig	Cup plant	49.520° N, 7.073° E	SE	370	8	Silt loam	4.96	16.09	1.81
	Maize	49.520° N, 7.073° E	SE	359	5	Silt loam	5.98	16.15	1.79
Remmesweiler	Cup plant	49.434° N, 7.119° E	NNW	285	6	Silt loam	4.25	16.69	1.73
	Maize	49.450° N, 7.136° E	N	305	3	Silt loam	6.41	17.88	1.88
Dörrenbach	Cup plant	49.440° N, 7.211° E	NNE	381	3	Silt loam	5.53	20.76	2.28
	Maize	49.439° N, 7.219° E	NNE	382	7.5	Silt loam	6.36	23.14	2.28
Fürth	Cup plant	49.425° N, 7.229° E	NNE	283	9	Silt loam	6.10	25.62	2.55
	Maize	49.427° N, 7.201° E	E	309	2.5	Sandy loam	5.63	17.06	1.63

1981). The collars (65 × 45 × 15.6 cm) for the white vented chambers (68 × 48 × 30 cm; PS-Plastic, Eching, Germany) were inserted 7.5 cm in the soil. Each chamber was equipped with an electric fan, vent tubes, a thermometer, and an inlet and outlet for gas sampling. Chambers were clamped on the collars with clips and sealed air-tight by rubber seals. Headspace air samples were drawn with a handheld electric air pump into 20-mL glass vials closed with a rubber septum. Before taking the sample, the vial volume was flushed 40 times by circulating headspace air through the vial. The closure time of each chamber for gas flux measurement was 1 h and air samples were taken at 0, 20, 40, and 60 min. At the same time as each gas sample was taken, the soil and chamber air temperatures were recorded.

Chambers in cup plant fields were placed between the rows (75 cm row spacing). In maize fields, chambers were placed between or over the seeding row (no fertilizer band placement below seeding row, 50–75 cm row spacing). When chambers had to be placed on the maize row (Remmesweiler, Dörrenbach, Fürth), seedlings were removed immediately after emergence. Collars had to be removed from the plots for harvest and tillage operations. Exceptionally, rainy weather conditions in 2019 delayed field operations leading to data gaps in the weekly measurements. To enable GHG measurement in the cup plant fields, single plants had to be removed around the collars. This caused a canopy gap with altered ground shading, which affected soil temperature and thus other soil-related processes. Therefore, in the 2020/2021 season, camouflage nets were placed above the collars to mimic canopy shading while allowing natural rain penetration.

Gas samples were analyzed for CO_2, N_2O, and CH_4 concentrations using a gas chromatograph (GC-2014; Shimadzu, Duisburg, Germany) equipped with an electron capture detector (ECD) and a flame ionization detector (FID), and connected to an autosampler (Greenhouse Workstation AS-210, SRI Instruments Europe GmbH, Bad Honnef, Germany). Four standard gases with concentrations from 300 ppb N_2O/350 ppm CO_2/1.4 ppm CH_4 to 3000 ppb N_2O/4000 ppm CO_2/5 ppm CH_4 in synthetic air were used for calibration. The precision of the GC was regularly tested by repeated measurement of standards with gas concentrations close to ambient, and the coefficient of variance (CV; $n = 10$) was always <2%.

2.3 | Soil analysis and soil data collection

Samples for determining soil mineral nitrogen (N_{min} = nitrate [NO_3^-] + ammonium [NH_4^+]) were taken weekly from 0- to 30-cm depth during GHG measurement using a Goettinger boring rod with an inner diameter of 1.8 cm (Nietfeld GmbH, Quakenbrück, Germany). N_{min} samples from 0 to 90 cm in three depths were taken twice a year, after harvest and at the beginning of the following growing season. Samples were stored at −20°C until extraction with 2 M KCl (1:4 [m/v]). The NO_3^- and NH_4^+ concentrations in the extractant were quantified using a photometric continuous flow analyzer (SEAL Analytical, Southampton, United Kingdom).

Soil moisture and temperature at 7.5 and 17.5 cm soil depth were measured continuously at each field using time domain reflectometry (TDR) sensors (5 TM, Meter Group, Munich, Germany) connected to a data logger (GP 2, Delta-T-Devices, Cambridge, United Kingdom).

Bulk density was determined in each year of the experiment at representative dates in the vegetation (8–10 weeks after tillage) period, as well as during winter. At each sampling, six undisturbed soil samples were collected using 100 cm^3 cylinders. Soil samples were dried at 105°C for 24 h and the masses of the soil cores determined. The water holding capacity (WHC) was determined as total pore volume minus the pore volume of pores >50 μm diameter, both determined by dewatering soil cores using overpressure.

2.4 | Biomass and biogas yield determination

Biomass yields were determined on three randomly chosen plots per field a few days prior to harvesting by the farmer. On each field, three 200-cm plant rows (row spacing: 50– or 75-cm) were harvested by hand at a cutting height of 12– to 15-cm. Thus, the sampled area was between 3 and 4.5 m^2 per plot. Fresh matter of the harvested biomass was weighed in the field using a hanging balance. For determination of dry matter contents, a representative aliquot (approx. 5 kg) of the harvested whole plants was taken to the laboratory immediately and chopped to chop lengths of between 3 and 7 mm using a blade shredder. The chopped biomass was dried at 105°C for at least 24 h. Dry matter

contents were then calculated from the water loss. Biomass yields on a dry matter basis were finally calculated by extrapolating the fresh matter yields to 1 ha and multiplying this by the dry matter contents. For determination of the N content of harvested biomass, plant material was oven-dried at 60°C, finely ground, and subsequently analyzed by gas chromatography after combustion at 1100°C using a EuroEA elemental analyzer (HekaTech, Wegberg, Germany). A further aliquot of each chopped (5–10 mm length) sample was immediately ensiled without any additives in vacuum bags for at least 8 weeks. Biogas batch assays with a duration of 50 days were performed in duplicates according to VDI Guideline 4630 (VDI, 2016) under mesophilic conditions (37°C) and a substrate-to-inoculum ratio of 0.40.

2.5 | Data analysis and statistics

The field experiment was divided into crop years 2019/2020 and 2020/2021 (hereinafter 2019/20 and 2020/21) from March to March. Furthermore, each crop year was separated into the growing season from the beginning of March to the end of September (214 days) and the winter season from October to the end of February (151 and 152 days) of the years 2019/20 and 2020/21.

Residual N_{min} content was determined by considering the first two samples in October or the first two samples after harvest for maize sites harvested in October. Furthermore, unless otherwise stated, N_{min} values depict the content in 0–30 cm soil depth.

The nitrogen use efficiency (NUE) in this study was calculated as the ratio of N applied as fertilizer divided by the N harvested with the biomass. Symbiotic N fixation, deposition, and mineralization were not considered in this approach. The productivities (land and fertilizer) of the cropping systems were calculated as input/output ratios.

GHG fluxes were quantified from the accumulation of gas concentrations in the headspace during the closure time of the chambers. From these gas concentrations, the mass concentrations were obtained to calculate the corresponding mass flow per area and time according to the ideal gas law. CO_2 fluxes were used as the quality parameter for the flux measurement. From the measured gas fluxes, cumulated annual and seasonal emissions were obtained with the trapezoidal rule. Further analysis and calculation of yield-related emissions were based on these cumulated emissions. The share of growing season N_2O emissions in annual seasonal emissions was calculated to describe the seasonal pattern of N_2O emissions.

The CH_4 uptake was not used for further calculations. Mineral upland soils are net CH_4 sinks (Le Mer & Roger, 2001) and it is known that, for example, soil properties are decisive for the sink strength of a soil. This implicates that distinct value cannot be assumed as baseline net flux. Therefore, a measured net CH_4 uptake cannot be identified as a sink or a source of CH_4 because no individual reference for the sink strength is available.

Energy yields (kWh_{el} per kg DM or ha) were derived from dry matter hectare yield (DMY) and corresponding specific methane yield (SMY). Silage, storage, and transport losses of 12% (Doehler et al., 2013) were assumed for yield calculations. Biomass yield-scaled emissions were calculated with harvested DMY without considering the above losses. Methane hectare yield (MHY) was calculated as follows:

$$MHY = DMY_{lc} \times VS \times SMY, \quad (1)$$

where DMY_{lc} is the loss-corrected (DMY – 0.12 × DMY) dry matter hectare yield, VS (volatile solids) is the organic dry matter content, and SMY is the specific CH_4 yield as standard liter (SL) CH_4 kg^{-1} VS. Gas yields were standardized to the conditions of 1013.25 hPa and 273.15 K.

The conversion of 1 m^3 of methane (m^3 CH_4) into electrical power (kWh_{el}) was calculated as follows:

$$kWh_{el} = m^3CH_4 \times 9.968 \times 0.38. \quad (2)$$

One m^3 of CH_4 has a calorific value of 9.968 kWh, while the typical efficiency for electrical power of a co-generation plant is 38%, which is equivalent to 3.8 kWh m^{-3} CH_4 (Doehler et al., 2013). In this study, kWh_{el} was used as the standardized energy output because the utilization efficiency of thermal energy varies between biogas plants.

CO_2-equivalent ($CO_{2\text{-eq}}$) fluxes were calculated from N_2O fluxes with a 100-year global warming potential of 265, according the fifth assessment report of the IPCC (Myhre et al., 2013).

R version 4.0.2 was used for all statistical analyses. Cumulated emissions were analyzed by fitting linear mixed-effect models. Site-pairs were integrated into the linear mixed-effects models as random effects (*lme4*; Bates et al., 2020) and cropping system and crop year were fixed effects. For the analysis of percentages, generalized linear models were fitted using a quasi-binomial distribution family and logit link function (Pinheiro et al., 2020). The assumption of normality and homoscedasticity was inspected graphically. Pairwise comparisons within the site-pairs were conducted with pairwise *t*-tests and Holm's *p*-value adjustment or pairwise comparison with the R package "*emmeans*" (Lenth et al., 2019). The N_2O fluxes were consistently log10-transformed for further analysis to reduce variance heterogeneity as far as possible. Unless otherwise stated, the results presented in this work are arithmetic mean ± 1 standard deviation.

3 | RESULTS

3.1 | Weather and soil conditions

Annual precipitation was 1174.4 and 873.8 mm for the 2019/20 and 2020/21 crop years, respectively. Therefore, the 2019/20 crop year received 143.4 mm more precipitation than the long-term annual mean, and was 34% wetter than the 2020/21 crop year, which received 157.2 mm less rain than the local long-term mean. Corresponding to the high annual precipitation, the 2019 growing season received 40% (143.7 mm) more rain than the 2020 growing season. Furthermore, the mean temperature in the 2019 growing season was 0.5°C lower than in the 2020 growing season (Figure 1). In the warmer, drier 2020 growing season, soil temperature followed the daily mean air

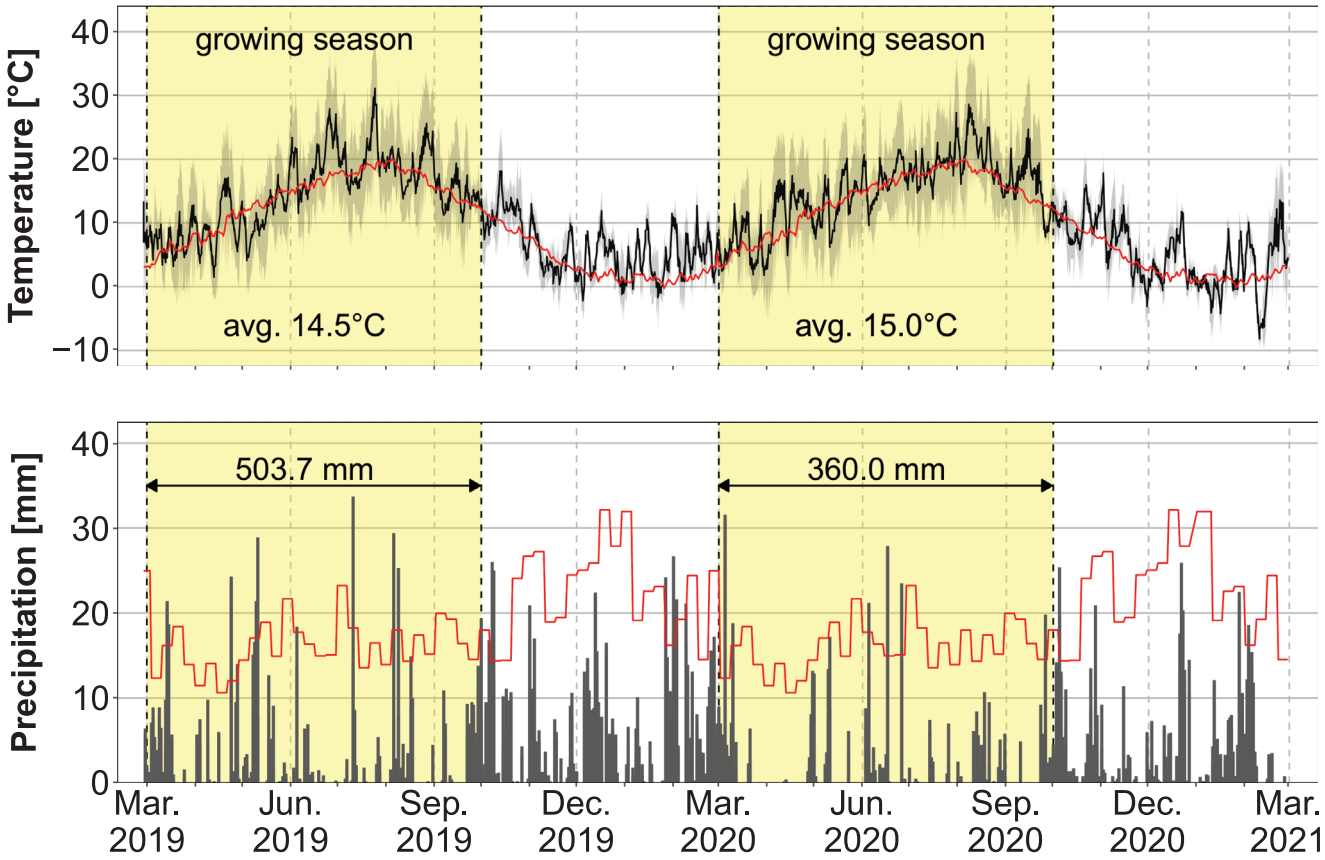

FIGURE 1 Mean daily temperature and the range between minimum and maximum daily temperature (black line and shaded area) with the red line depicting the 30-year average daily temperature. Daily precipitation and 30-year annual average of weekly precipitation (red) over the two experimental years (DWD, 2021)

temperature closely, especially under maize canopies. In cup plant fields, during the wetter 2019 growing season, soil temperatures appeared to be slightly lower than mean air temperature during phases of high summer temperatures, and also lower than the soil temperature in the corresponding maize fields.

In both years with the onset of more frequent rain events in the fall, soil moisture increased up to field capacity and remained high. Therefore, soils were temporarily waterlogged from October to late March in both years of the experiment. In January and February 2019, 126.6 mm rain fell, while in the same 2 months in 2020 the amount was higher (294.3 mm). However, the first half (March to mid-June) of the 2019 growing season received around 100 mm more rain than the same period in the 2020/21 crop year. Therefore, the continuous decline in soil moisture in the subsequent months toward its minimum was delayed in spring 2019. The lowest soil moisture of around 42% and 36% of the soil WHC was observed in September in 2019 and 2020, respectively (Figure S1). The depletion of plant-available water did not differ ($p = 0.36$) between the two biomass crops, but the year had a significant effect ($p = 0.01$) on mean plant-available water.

With a mean soil pH of 5.21 ± 0.79 and 6.10 ± 0.36 in cup plant and maize, respectively, soil pH tended ($p = 0.2$) to be lower in cup plant fields. SOC did not differ ($p = 0.65$) between cup plant fields (19.79 ± 4.41 g kg^{-1}) and maize fields (18.56 ± 3.14 g kg^{-1}) (Table 1).

3.2 | Soil mineral nitrogen

In 2019/20, soil mineral nitrogen (N_{min}) contents did not differ ($p = 0.85$) between the two crops, and during the 2019 growing season N_{min} ranged between 30 and 70 kg N ha^{-1} (Table S2). N_{min} contents in the following 2019/20 winter season tended ($p = 0.06$) to be around 50% lower than in the growing season.

In the 2020 growing season, N_{min} contents in maize fields (97.6 ± 83.9 kg N ha^{-1}) were significantly ($p < 0.001$) higher than in cup plant fields (31.9 ± 19.5 kg N ha^{-1}). However, the N_{min} contents of the two crops were comparable in both winter seasons.

N_{min} levels in maize remained relatively high after fertilization and throughout the growing season, especially in 2020, and decreased with winter precipitation. At the maize site in Gronig, N_{min} > 200 kg N ha^{-1} was frequently measured throughout the 2020 growing season (Figure 2). However, N_{min} at the end of the winter (January and February) was consistently low in both systems (16.1–42.9 kg N ha^{-1}), particularly the NO_3^- concentrations (1.1–11.1 kg N ha^{-1}), although cup plant in Dörrenbach and Fürth was fertilized early in February. The reduction over the winter in N_{min} in 0–30 cm in maize did not result in higher N_{min} contents in deeper soil (Figure S2). Generally, N_{min} content in maize was always higher ($p < 0.001$) in the growing season than in the winter season, but this was not observed in cup

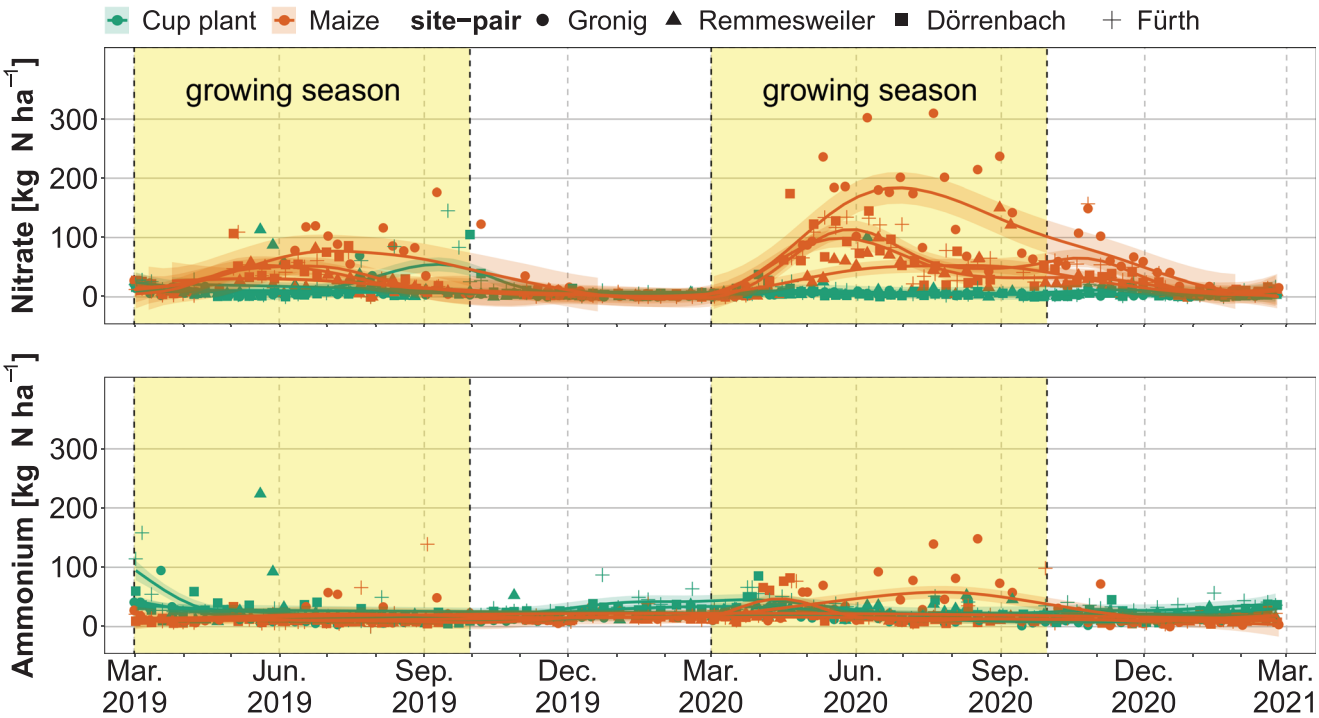

FIGURE 2 Weekly nitrate, ammonium, and total soil mineral nitrogen (N_{min}: NO_3^--N and NH_4^+-N) data (0–30 cm depth) and plotted line from a generalized additive model (GAM) ± 1 standard error to visualize the N_{min} dynamic over the time course of the two experimental years and each crop and site-pair

plant fields ($p = 0.35$). N_{min} levels in cup plant decreased shortly after fertilization and remained low. Residual N_{min} content after harvest in 2019/20 was comparable between the two crops at around 25 kg N ha^{-1}. In the dry 2020/21 crop year, however, residual N_{min} in maize (58.2 ± 21.5 kg N ha^{-1}) was significantly higher ($p = 0.05$) than in cup plant (19.7 ± 11.5 kg N ha^{-1}). Overall, the residual N_{min} content was positively correlated with applied N ($r = 0.72$, $p = 0.002$).

The differences in N_{min} levels between crops and seasons were predominately caused by the differing NO_3^- contents (Figure 2), since NH_4^+ did not differ significantly between seasons and crops. In both years, this resulted in a higher ($p < 0.001$) fraction of NH_4^+-N in cup plant (76.7% ± 18.6%) than in maize (45.7% ± 28.4%), with this fraction lowest in maize 2020/21.

3.3 | CH_4 fluxes

Mean soil methane (CH_4) uptake was 1.35 ± 0.53 kg C ha^{-1} $year^{-1}$ and 0.64 ± 0.19 kg C ha^{-1} $year^{-1}$ in cup plant and maize, respectively. However, soil-derived CH_4 emission events occurred frequently in cup plant and maize fields over the winter, where soils were nearly saturated. The highest observed emission rate was 52.5 μg C m^{-2} h^{-1} (Figure S3). Pairwise *t*-tests revealed that cup plant soil had significantly ($p < 0.01$) higher methane uptake than maize soil, except at the Gronig site ($p = 0.09$). Annual CH_4 net uptake did not differ ($p = 0.42$) between 2019/20 and 2020/21. The difference in uptake between cup plant and maize was greater in the growing season ($p < 0.001$) than in the winter season ($p = 0.02$). Hence, uptake in both crops was significantly higher in the growing season than in the winter. Furthermore, the difference between growing season and winter season was more pronounced in cup plant, especially in Dörrenbach and Fürth (Figure S3).

3.4 | N_2O emissions

Over the 2 years of observation, maize (4.23 ± 4.26 kg N ha^{-1} $year^{-1}$) had significantly higher ($p < 0.001$) annual N_2O emissions than cup plant (0.62 ± 0.39 kg N ha^{-1} $year^{-1}$). Furthermore, analysis of variance showed that there was a significant year effect ($p = 0.02$), with lower N_2O emissions in the second year, and a significant interaction of year and crop effects ($p = 0.02$).

In 2019/20, cumulated N_2O emissions were within the range of 1.35 6.07 kg N ha^{-1} $year^{-1}$ and 0.70–1.20 kg N ha^{-1} $year^{-1}$ from maize and cup plant fields, respectively (Table 2). Thus, in 2019/20 mean cumulated N_2O emissions from maize were on average 4.4 times higher than from cup plant ($p < 0.001$). At three of the sites, maize had significantly ($p = 0.04$–0.003) higher annual emissions; the exception was Dörrenbach where the annual N_2O emissions did not differ ($p = 0.5$) between the two cropping systems. In cup plant in the wetter year of 2019/20, more ($p < 0.001$) N_2O was emitted during the growing season (0.77 ± 0.38 kg N ha^{-1} 214 day^{-1}) than during the winter season (0.08 ± 0.06 kg N ha^{-1} 152 day^{-1}). In maize, N_2O emissions during the growing season (2.77 ± 1.53 kg N ha^{-1} 214 day^{-1}) were also

TABLE 2 Cumulated growing season, winter season, and annual N_2O emission in the 2019/2020 and 2020/2021 crop years at each site-pair, ± 1 standard deviation ($n = 4$)

		2019/2020			2020/2021		
Site	Crop	Growing season (kg N ha^{-1} 214 day^{-1})	Winter season (kg N ha^{-1} 152 day^{-1})	Annual (kg N ha^{-1} year^{-1})	Growing season (kg N ha^{-1} 214 day^{-1})	Winter season (kg N ha^{-1} 151 day^{-1})	Annual (kg N ha^{-1} year^{-1})
Gronig	Cup plant	0.67 ± 0.24	0.03 ± 0.02	0.70 ± 0.25	0.36 ± 0.21	0.13 ± 0.04	0.49 ± 0.19
	Maize	4.24 ± 0.38	1.83 ± 0.21	6.07 ± 0.56	2.86 ± 0.66	11.09 ± 3.43	13.95 ± 3.58
Remmesweiler	Cup plant	1.05 ± 0.38	0.15 ± 0.07	1.20 ± 0.40	0.21 ± 0.09	0.05 ± 0.04	0.26 ± 0.12
	Maize	2.07 ± 0.15	0.35 ± 0.25	2.43 ± 0.13	1.93 ± 1.16	0.47 ± 0.29	2.40 ± 1.15
Dörrenbach	Cup plant	0.66 ± 0.50	0.05 ± 0.03	0.71 ± 0.50	0.31 ± 0.37	0.09 ± 0.02	0.41 ± 0.36
	Maize	1.15 ± 1.25	0.21 ± 0.33	1.35 ± 1.57	0.48 ± 0.14	0.79 ± 0.38	1.27 ± 0.51
Fürth	Cup plant	0.69 ± 0.37	0.08 ± 0.05	0.77 ± 0.37	0.32 ± 0.16	0.11 ± 0.05	0.43 ± 0.18
	Maize	3.62 ± 1.43	0.94 ± 0.60	4.56 ± 0.86	1.23 ± 0.82	0.57 ± 0.03	1.81 ± 0.84

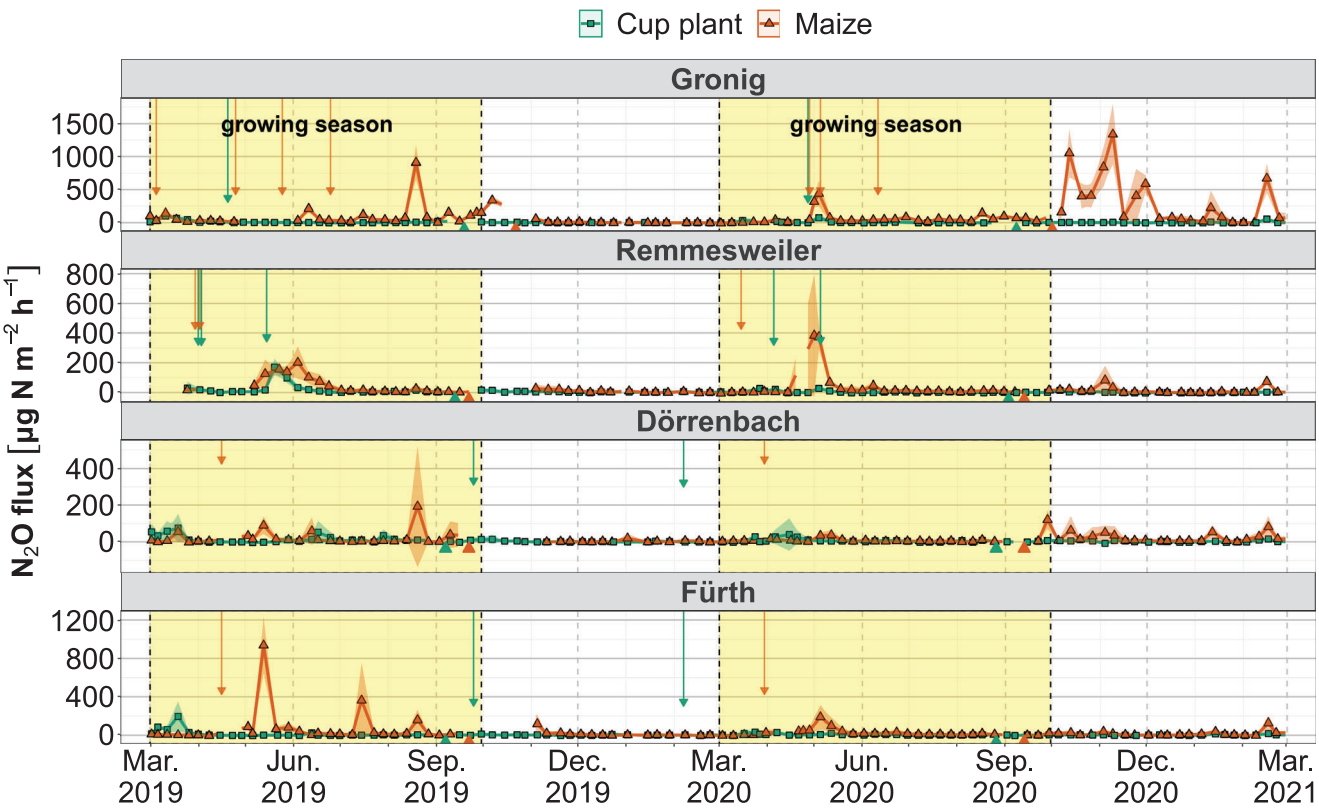

FIGURE 3 Nitrous oxide (N_2O) fluxes from cup plant and maize fields at each site-pair over the two experimental years. Shaded bands depict ± 1 standard deviation ($n = 4$). Arrows indicate fertilizer application and triangles indicate harvesting date. Yellow shaded areas mark the growing seasons. Note different *y*-axis scales

significantly higher ($p < 0.001$) than in the winter season (0.83 ± 0.74 kg N ha^{-1} 152 day^{-1}). However, winter emissions in maize were around 10 times higher than in cup plant ($p < 0.001$). In the 2019/20 season, the growing season contributed 89.7% ± 9.4% and 80.6% ± 11.6% to annual N_2O emissions in cup plant and maize, respectively. Therefore, most of the annual N_2O emissions occurred over the summer in several emission events (Figure 3).

In the 2020/21 season, cumulated N_2O emissions ranged between 1.27 and 13.95 kg N ha^{-1} year^{-1} and from 0.26 to 0.49 kg N ha^{-1} year^{-1} from maize and cup plant fields, respectively. Similarly, to the previous season, in 2020/21 maize fields exhibited significantly higher ($p = 0.01$ to <0.001) N_2O emissions than the corresponding cup plant fields, except at the Dörrenbach site ($p = 0.065$). In 2020/21, emissions from the maize field in Gronig were on average more than 7.5 times higher than emissions from the other maize fields (Table 2). Therefore, in 2020/21 this exceptionally high emission in Gronig greatly influenced the mean cumulated N_2O emissions from maize fields (4.86 ± 5.71 kg N ha^{-1} year^{-1}), resulting in emissions more than 12

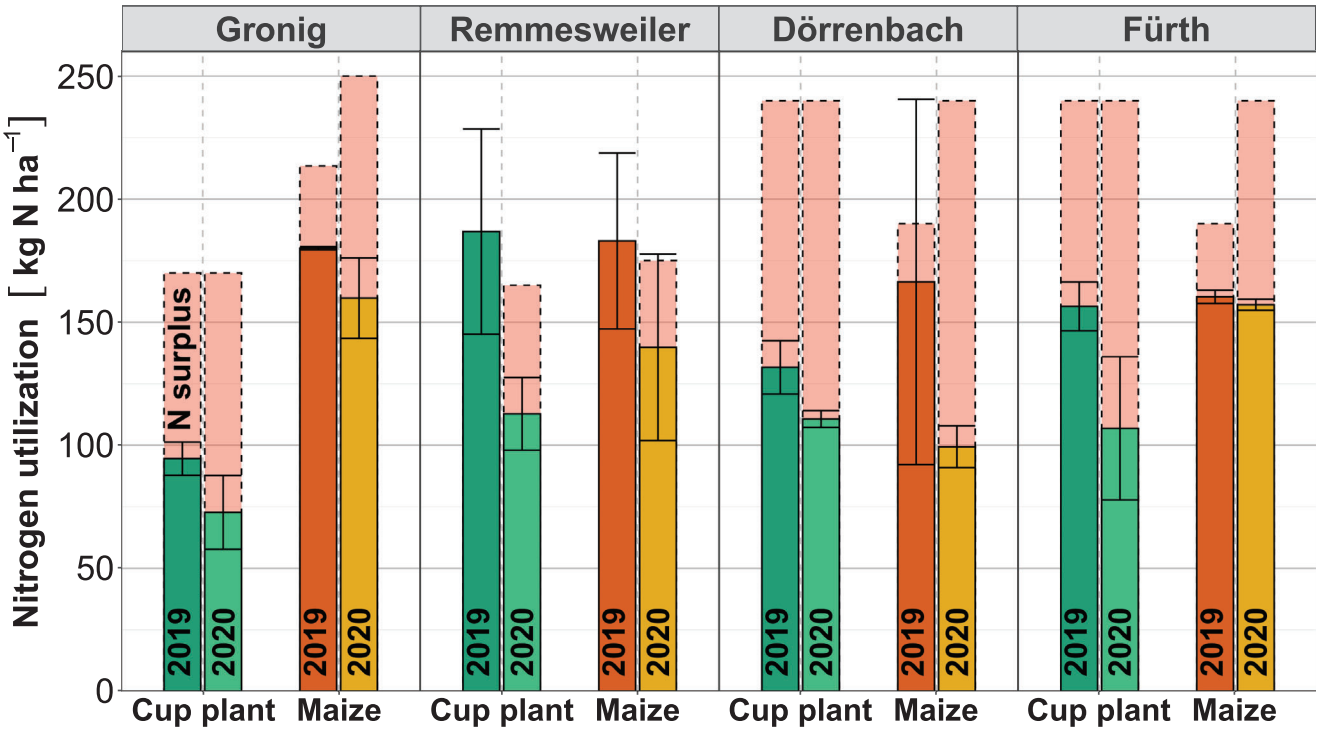

FIGURE 4 Nitrogen (N) exported in harvested cup plant and maize biomass and site-specific N fertilization in 2019 and 2020 at each site-pair. The difference between applied N and exported N represents the N surplus (red). Error bars depict ± 1 standard deviation ($n = 3$).

times higher ($p < 0.001$) than those from cup plant fields (0.40 ± 0.22 kg N ha^{-1} year^{-1}). In contrast to 2019/20, only small emissions occurred during the 2020/21 growing season, except for an emission event induced by fertilization (Figure 3). Mean growing season emissions were 0.30 ± 0.22 and 1.63 ± 1.15 kg N ha^{-1} 214 day^{-1} from cup plant and maize fields, respectively. The lower emissions during the growing season in the drier crop year of 2020/21 resulted in comparable ($p = 0.12$) emissions from the growing and winter seasons. Starting with the rainfall after the dry summer in 2020, emission events occurred at the maize sites during the winter season (Figure 3). However, only at the maize site in Gronig, where the residual N_{min} content after harvest was relatively high, were N_2O emissions in the winter season significantly higher ($p < 0.001$) than in the growing season. Mean winter season emissions were 0.10 ± 0.04 and 3.23 ± 4.93 kg N ha^{-1} 151 day^{-1} from the cup plant and maize fields, respectively. Therefore, the emitted N_2O in the growing season constituted only 69.3% ± 21.6% and 50.5% ± 25.2% of total annual emissions from cup plant and maize, respectively, which for both crops was significantly ($p < 0.001$) lower than in 2019/20.

3.5 | Biomass, nitrogen, and biogas yields

3.5.1 | Biomass

The biomass yields reflected the previously described pattern between the two seasons. In the wetter year, 2019/20, yields were 15.6 ± 3.8 and 16.6 ± 3.1 Mg DM ha^{-1} in cup plant and maize, respectively (Table S1). In the drier year, 2020/21, yields were 13.1 ± 2.7 and 14.0 ± 2.4 Mg DM ha^{-1} in cup plant and maize, respectively. Thus, dry matter yields were higher ($p = 0.002$) in the 2019/20 crop year than in the 2020/21 crop year. However, dry matter yields between cup plant and maize did not differ ($p = 0.24$) in the 2 years.

3.5.2 | Nitrogen

The mean N content of harvested maize biomass (10.2 ± 1.0 g N kg^{-1} DM) was higher ($p < 0.001$) than that of cup plant biomass (8.5 ± 1.8 g N kg^{-1} DM). This resulted in more exported N in maize (Figure 4) and lower NUE in cup plant ($p = 0.001$). N concentration ($p = 0.01$) and NUE ($p < 0.001$) exhibited a clear year effect toward less N utilization in the drier season. The higher yielding conditions in 2019/20 resulted in significantly ($p = 0.004$) higher N surpluses in cup plant (+57.2 ± 62.2 kg N ha^{-1}) than in maize (+12.8 ± 45.8 kg N ha^{-1}). In the drier season of 2020/21, N surplus showed similar tendencies, but N surpluses in cup plant (103.0 ± 37.2 kg N ha^{-1}) were not significantly ($p = 0.57$) higher than in maize (87.3 ± 43.0 kg N ha^{-1}).

3.5.3 | Biogas

In contrast to biomass yields, the specific methane yield (SMY) from anaerobic digestion per organic dry matter (VS) of cup plant biomass (296.8 ± 50.6 SL kg^{-1} VS) was significantly ($p < 0.001$) lower than from maize biomass (383.3 ± 37.4 SL kg^{-1} VS) and it was not affected by the drought in 2020/21 ($p = 0.52$).

TABLE 3 Biomass and energy yield-scaled GHG emissions. CO_{2eq} was calculated solely from measured field-derived N_2O emissions with a global warming potential (GWP) of 265. Area and fertilizer productivity of cup plant and maize in the 2019/2020 and 2020/2021 seasons, ± 1 standard deviation ($n = 3$)

Crop	Season	Emitted g $CO_{2\text{-eq}}$ kg DM harvested^{-1}	Emitted g $CO_{2\text{-eq}}$ kWh^{-1}	Emitted N/applied N	kWh m^{-2}	kWh kg N^{-1}
Cup plant	2019/20	23.0 ± 11.6 b	26.1 ± 12.8 b	0.4 ± 0.2 b	1.4 ± 0.3 b	74.4 ± 27.4 bc
	2020/21	13.6 ± 7.4 c	15.1 ± 8.2 c	0.2 ± 0.1 c	1.1 ± 0.2 b	56.1 ± 9.1 c
Maize	2019/20	91.9 ± 40.2 a	74.5 ± 33.3 a	1.9 ± 0.9 a	2.0 ± 0.4 a	109.8 ± 28.8 a
	2020/21	145.0 ± 59.7 a	114.8 ± 48.8 a	2.1 ± 2.2 a	1.8 ± 0.4 a	79.4 ± 17.6 b

This resulted in higher ($p < 0.001$) electrical power yields per ha^{-1} from maize with 20.0 ± 4.0 and 17.8 ± 3.6 MWh ha^{-1} than cup plant with 13.9 ± 3.1 and 11.3 ± 2.3 MWh ha^{-1} in 2019/20 and 2020/21, respectively (Table 3). There was a slight year effect ($p = 0.01$) on the electrical power yield with on average −2.4 MWh ha^{-1} less in season 2020/21.

3.6 | Yield and fertilizer related emissions and factor productivity

In the 2019/20 crop year, cup plant had 0.16 ± 0.14 kg N_2O-N Mg^{-1} DM lower ($p < 0.001$) yield-related emissions than maize systems. In the drier 2020/21 crop year, this difference (0.30 ± 0.41 kg N Mg^{-1} DM; $p < 0.001$) was even more pronounced. Based on a comparison of estimated marginal means, cup plant fields emitted significantly ($p < 0.001$) less direct N_2O per produced kWh in both the 2019/20 and 2020/21 crop years than maize fields (Table 3). In the 2020/21 crop year, energy and DM output-related emissions were greatly affected by the high N_2O emissions at the maize site in Gronig. Excluding this site-pair in the drier 2020/21 crop year, output-related emissions from cup plant fields were, however, still lower than from maize fields.

Fertilizer-related emissions (emitted N_2O-N per applied N) were on average 0.3% ± 0.2% and 2.0% ± 1.7% in the cup plant and maize systems, respectively (Table 3). Thus, the emissions per applied kg N were higher ($p < 0.001$) in maize. The crop × year interaction effect ($p = 0.05$) indicated that fertilizer-related emissions in the drier year 2020/21 were only lower in cup plant, not in maize (Table 3).

In contrast to yield-scaled emissions, the productivity of used production area and applied N fertilizer tended to be lower for cup plant (Table 3). To produce 1 MWh using cup plant alone as the substrate, 756.5 ± 153.1 and 916.0 ± 165.3 m^2 would have been needed in the 2019/20 and 2020/21 crop year, respectively. Maize required significantly ($p < 0.001$) less acreage to produce the same amount of electrical power (524.7 ± 134.2 and 586.5 ± 128.9 m^2 in the 2019/20 and 2020/21 crop year, respectively). Consequently, area productivity of cup plant was 33.6% ± 27.6% lower. A similar effect was observed regarding the productivity of applied N fertilizer, where less ($p < 0.001$) power could be generated per kg N applied in cup plant. The N productivity exhibited a more pronounced year effect ($p < 0.001$) than the area productivity ($p = 0.01$), with lower productivity in the drier season of 2020/21.

4 | DISCUSSIONS

The perennial cup plant system had on average only 14.7% ± 20.6% of the annual N_2O emissions of maize; however, both crops received similar amounts of N fertilizer at around 200 kg N ha^{-1}. The mean annual N_2O emission from maize with 4.2 ± 4.3 kg N_2O-N ha^{-1} year^{-1} was in the range of 3.6–6.6 kg N_2O-N ha^{-1} year^{-1}, as reported in other studies on fine-textured soils and comparable N-application rates (162–240 kg N ha^{-1}) (Gauder et al., 2012; Van Groenigen et al., 2004). No information on annual N_2O emissions from cup plant fields is as yet available in the literature.

A suitable agronomic alternative to cup plant at these experimental sites could be grassland, since it is perennial and also commonly serves as biogas substrate, and can be assumed to be superior even to cup plant as regards soil-preserving benefits (Peeters, 2009; Weiland, 2006). Annual N_2O emissions between 0.65 and 2.25 kg N_2O-N ha^{-1} year^{-1} have been reported from grassland under soil and climatic conditions and N rates (120–240 kg N ha^{-1}) comparable to those in the present field experiment (Cardenas et al., 2019; Hargreaves et al., 2021; Jungkunst et al., 2006). Thus, emissions from grassland tended to be slightly higher than the emissions measured at the cup plant fields (0.62 ± 0.39 kg N_2O-N ha^{-1} year^{-1}). However, in contrast to a single application in cup plant, multiple fertilizer applications are common practice in grassland management, which increases the risk of high emission during hot moments after fertilization.

Although the experimental sites with their fine-textured soil and imperfect drainage were prone to temporal water-logging and thus reducing conditions, both cropping systems were net CH_4 sinks in the growing and winter season. However, cup plant fields exhibited a 0.71 ± 0.50 kg C ha^{-1} year^{-1} (26.46 ± 18.81 kg $CO_{2\text{-eq}}$ ha^{-1} year^{-1}) higher methane oxidation ($p < 0.001$) than maize fields. The observed CH_4 uptake of 0.41–2.04 kg C ha^{-1} year^{-1} from maize and cup plant cropping in this study was in the range (0.02–2.3 kg C ha^{-1} year^{-1}) of the reported CH_4 uptake from arable land (Hütsch, 2001) and annual and perennial biomass cropping systems (Gauder et al., 2012; Walter et al., 2015) in central Europe.

Dry matter yields of maize in this experiment ranged from 10.2 to 21.0 Mg ha^{-1} year^{-1}, with lower ($p = 0.002$) yields in the drier season (2020/21), which were comparable with yields of between 11.7 and 17.1 Mg ha^{-1} year^{-1} reported from other fine-textured sites and similar N rates (Rochette, Angers, Chantigny, & Bertrand, 2008; Rochette, Angers, Chantigny, Gagnon et al., 2008; Van Groenigen et al., 2004). Therefore, measured yields from the low mountain range were lower than yields from areas more suitable for silage maize production (15.2–27.4 Mg DM ha^{-1} year^{-1}) (Brauer-Siebrecht et al., 2016; Herrmann et al., 2013). In contrast to maize, cup plant yielded 9.4–21.6 Mg DM ha^{-1} year^{-1}, which is comparable to yields found in the literature (7.2–22.5 Mg DM ha^{-1} year^{-1}) by Gansberger et al. (2015) at many different sites and N application rates.

4.1 | N_2O emission from fine-textured marginal sites

The experimental sites can be described as marginal cropland for maize production since cultivation is somewhat challenging (Blanco-Canqui, 2016) due to the fine soil texture (>20% clay and >50% silt), slope, soil water condition, and climatic conditions (>1000 mm year^{-1}). These fine-textured and slower-draining soils are more frequently exposed to anoxic conditions due to reduced gas diffusivity, and thus favor N_2O emissions from denitrification compared with soils with a coarser texture (Rochette, Angers, Chantigny, & Bertrand, 2008; Rochette, Angers, Chantigny, Gagnon et al., 2008). This explains the high N_2O emission potential from denitrification of the experimental sites with up to a maximum of 14 kg N_2O-N ha^{-1} year^{-1} at plot scale, which was 5.6% of the applied N. Under these conditions, in both experimental years, the cultivation of cup plant caused significantly lower soil-derived annual N_2O emissions than maize, and thus supported the first hypothesis. Based on field-derived emissions, cup plant provides a greater N_2O mitigation potential compared with maize at these marginal sites with fine-textured soils.

The soil moisture monitored in the individual cup plant and maize field (Figure S1) did not explain the N_2O fluxes well. However, emissions events, especially in the 2019/20 season, occurred frequently throughout the growing season after heavy precipitation, so-called hot moments (Groffman et al., 2009; Krichels & Yang, 2019). In contrast to Krichels and Yang (2019) and Parkin (2008), in this study hot moments could be observed after smaller rain events or cumulative rain of >10 mm on consecutive days, where soil moisture was far below field capacity. Due to their pore size distribution, fine-textured and polydispersed soils are more prone to water blockage of the uppermost pores, which also last longer than in coarser and faster draining soils (Rochette, 2008; Saxton et al., 1986). Therefore, rain events favor denitrification due to hypoxia and cause N_2O accumulation beneath this water blockage. The accumulated N_2O is subsequently released with a time delay after the actual rain event (Maier et al., 2019; Maier et al., 2011). Furthermore, anoxic conditions are aggravated by a high level of microbial activity, such as that found in the topsoil and at warmer temperatures, for example, in summer. Thus, we assumed that smaller rain events cause higher N_2O emissions from these fine-textured soils compared with coarser soils due to frequently interrupted soil–air gas exchange. However, the effect of water-induced surface pore blockage on N_2O emissions could not be adequately addressed in this study due to the limited temporal resolution of weekly GHG measurements. Nonetheless, we assume that rain events under the abovementioned conditions might explain the occurrence of N_2O emission events at these sites more precisely than soil moisture alone (Rowlings et al., 2015), regardless of the cropping system (Figure S1).

4.2 | N_{min} dynamic and N_2O emissions

The two cropping systems exhibited a clearly distinguishable pattern in N_{min} contents. In the maize system, the N_{min} contents during the growing season tended to be higher (Figure 2; in 2019/20: $p = 0.84$; in 2020/21: $p = 0.004$) than in the cup plant system, while N_{min} contents over the winter did not differ. The N requirement of cup plant and maize without subtracting initial N_{min} before fertilization is 130–160 and 200 kg N ha^{-1}, respectively (BMEL, 2017; Gansberger et al., 2015). However, both crops received ≈200 kg N ha^{-1} and were thus overfertilized. In particular, sites that have been cultivated continuously with maize and amended with organic fertilizer over a long period (Dörrenbach and Fürth) potentially exhibit an N supply exceeding crop demand due to excessive N mineralization (Gutser et al., 2005; Schröder et al., 2005). This increases the risk of residual or re-mineralized N losses due to leaching and denitrification after crop uptake ceases.

Interestingly, the N_{min} in cup plant diminished relatively quickly after fertilization, and remained at a low level <50 kg N ha^{-1} (Figure 2). This rapid and more complete depletion of N_{min} pools is typical of established perennials that are capable of utilizing available N immediately as previously reported for cup plant (Grunwald et al., 2020) and other perennial biomass crops (Pugesgaard et al., 2015).

In contrast to cup plant, maize did not deplete the N_{min} pool as rapidly, resulting in higher N_{min} availability during the growing seasons and in the fall, especially in the drier season of 2020/21 (Figure 2). At the time of fertilization, N acquisition of juvenile maize stands is insufficient to deplete all available N pools because of limited rooting of the interrow soil, especially under conditions with limited mass flow to the plant rows (Engels and Marschner, 1995). High NO_3^- contents over a longer period with high soil temperature substantially increase the risk of denitrification losses (Butterbach-Bahl et al., 2013), since Q_{10} values of between 2 and 6.2 are reported for denitrification (Abdalla et al., 2009; Phillips et al., 2015). Later in the season, N uptake reduced, presumably due to a pronounced summer drought. It has been reported that N uptake is increased with more available water and vice versa (Kim et al., 2008), which is consistent with the observation of lower N_{min} in the more humid year 2019/20 (Figure 2). The low N uptake of maize in the dry year 2020/21 resulted in high residual N_{min} in the fall, which in turn coincided with an increased share of winter emissions in annual emissions.

Moreover, a higher ($p = 0.02$) estimated mineralization could be observed at maize sites, since mean N_{min} during the growing season

in part substantially exceeded the difference between exported N and applied N. Higher mineralization rates in maize than in cup plant soil at the sites in Gronig were also shown in an incubation study (Kemmann et al., 2021). It has been reported that under no-till or reduced tillage management C and N turnover is lower than in conventionally managed soil (Kristensen et al., 2000; Six et al., 1999; Six et al., 1998) because tillage operations cause the release of previously protected C and N sources by breaking up soil aggregates and increasing soil aeration. In addition to the higher N_{min} content in maize, the fraction of NO_3-N at total N_{min} was also higher ($p < 0.01$) in maize than in cup plant. This indicates that in the maize soil processes of the N-cycle were dominant, which changed the N_{min} stoichiometry toward more NO_3^-. It has been shown that in cup plant soil more N was transferred from the NO_3^--pool into the less mobile NH_4^+-pool than in maize soil (Kemmann, 2022). This t makes the maize system more prone to N losses due to denitrification and leaching than the perennial system (Figure S2).

Although N_{min} tended ($p = 0.4$) to be higher and N_2O emissions were higher ($p < 0.001$) in maize than in cup plant, N_{min} and N surplus were not good predictors for N_2O fluxes. Only in maize 2019/20 did NO_3^- content clearly correlated positively ($R^2 = 0.44$) with N_2O emissions. In the 2020/21 crop year and in cup plant in both years, NO_3^- content was not obviously controlling N_2O emissions. According to Aulakh et al. (1992), NO_3^- content under natural conditions is not the single most important regulator of denitrification due to the complexity of factors controlling denitrification. This supports the observation that the soil texture and interrelated physical properties (gas diffusivity) at these marginal sites, which are susceptible to anoxic events, were more important for the control of N_2O emissions than N_{min} or NO_3^- content. This is consistent with Rochette, Angers, Chantigny, & Bertrand (2008) and Rochette, Angers, Chantigny, Gagnon, et al. (2008), who point out that soil aeration is a main driver of soil-derived N_2O emissions on fine-textured soil and that available NO_3^- increases the risk of N_2O emissions. The high NO_3^- contents, which coincided with the high N_2O emissions at the maize site in Gronig, might illustrate the described risk of NO_3^- and associated N_2O emissions.

Hence, the year-round low N_{min} content in cup plant cropping reduced the risk of N_2O emissions, although the crop received comparable amounts of N input. This therefore supports the second hypothesis that the application of N and its management in cup plant causes less N_2O emissions because available N_{min} and especially NO_3^- pools would be kept low under the perennial crop. However, less of the applied N was exported with the biomass (low NUE) and thus remained unaccounted for in the system or was lost to the environment. Therefore, fertilizer strategies are needed that are optimized with regard to the crop N demand.

4.3 | Input- and output-related emissions

Unlike other studies comparing perennial biomass crops, that is, miscanthus and willow, and annual crops (Gauder et al., 2012; Walter et al., 2015), the higher yield-scaled emissions of silage maize in this experiment could not be explained by higher N rates, since the N rates for cup plant and maize were almost identical. Compared with senescent harvested perennial biomass crops (miscanthus, switchgrass) or woody short rotation coppices (willow, poplar), green harvested cup plant exports more nutrients (Gansberger et al., 2015; Ruf & Emmerling, 2021; Ustak & Munoz, 2018). Therefore, cup plant requires a relatively intensive nutrient management, comparable with that of annual biomass crops (Ruf &Emmerling, 2021), and consequently fertilizer-induced emissions are also inevitable in cup plant cropping.

In addition, the applied N in cup plant cropping was apparently used less efficiently (NUE; $p = 0.001$) than in maize cropping. This could be a result of the lower biochemical NUE of C_3 crops compared with C_4 crops (Brown, 1978; Sage et al., 1987). However, due to the harvest of green biomass, retranslocation of N into belowground biomass at the time of harvest is expected to be less pronounced than in senescent harvested perennial crops such as miscanthus, which should increase the apparent fertilizer N recovery in cup plant. Furthermore, the longer phase of N uptake of perennial crops compared with annual crops improves N acquisition from fertilizer, and especially mineralization (Grunwald et al., 2020; Pugesgaard et al., 2015), which should also increase apparent N recovery. However, cup plant cropping tended to have higher ($p = 0.004$–0.57) N surpluses (only considering fertilizer N minus harvested N). A higher N balance in cup plant (−20 kg N ha^{-1}) has also been observed by Grunwald et al. (2020) using comparable N rates in maize (180 kg N ha^{-1}) and cup plant (170 kg N ha^{-1}). In other studies, lower or similar N balances have been found in perennial crops receiving lower N rates than annual crops (Cadoux et al., 2014; Smith et al., 2013). This might indicate that the lower N productivity in this study was due to a major unaccounted-for N loss. Since, these hydromorphic and slowly draining soils were completely saturated below 17.5 cm several times over winter, providing perfect anaerobic conditions for denitrification losses as N_2 (Well et al., 2003; Well et al., 2005). Furthermore, higher N_2O reduction rates were reported from cup plant than maize soil (Kemmann, 2022). However, in contrast to maize, N_{min} from cup plant fields (0–90 cm; Figure S2) were low and did not show excessive N losses suggesting that high rates of denitrification did not occur over the winter. Therefore, it is likely that the N_2O emissions measured over two years in this field experiment and due to the denitrification losses observed in two incubation studies (Kemmann, 2022; Kemmann et al., 2021), N_2 and N_2O emissions from the top soil do not explain the entire gap in the N balance of cup plant. Potential explanations for the unaccounted-for N loss in cup plant therefore might be denitrification in the subsoil, ammonia losses (fertilizer incorporation not common practice), surficial run-off, N immobilization into SOM, or N fixation in belowground and weed biomass (Lemus & Lal, 2005). However, since none of the aforementioned N loss paths were quantified, further investigations are needed in order to understand the fate of N in cup plant production.

One unit of harvested maize biomass had on average 549% ± 191% more ($p < 0.001$) field-derived emissions ($CO_{2\text{-eq}}$) than one unit of cup plant biomass due to its N_2O emissions. Considering the higher ($p < 0.001$) specific methane yield (SMY) of maize, the difference in emissions of produced electrical power output between the two

cropping systems narrowed compared with the difference in biomass yield-scaled emissions (Table 3). However, the electrical power that can be gained from 1 ha of cup plant was only 34% less than that from maize ($p < 0.001$). Hence, in order to generate the same amount of energy as from 1 ha of maize, the energy production solely with cup plant biomass would still cause less direct field-derived GHG emissions per unit energy produced. A 34% increase in cup plant production area still only produces 19% (35% if the maize site in Gronig with the extremely high N_2O emissions is excluded) of the emissions from 1 ha of maize.

Therefore, the third hypothesis, that cup plant cropping at these marginal sites causes less soil-borne N_2O emissions on an area and yield base, could be confirmed. However, in order to evaluate the GHG mitigation potential of replacing maize with cup plant in low mountain ranges, the associated risk of emissions due to land-use changes must be taken into account (Searchinger et al., 2009). To do so, the difference in productivity of the alternative crop could be used as a measure of the opportunity cost of replacing maize.

In this study, the productivity of arable land (MWh ha^{-1}; $p < 0.001$) and fertilizer (NUE; $p = 0.001$) was higher in the maize system. Pivotal to the greater area productivity was the higher SMY ($p < 0.001$) of maize (383.3 ± 37.4 SL kg^{-1} VS) compared with cup plant (296.8 ± 50.6 SL kg^{-1} VS), rather than biomass yield ($p = 0.24$). However, the observed SMY of cup plant was slightly higher than previously reported in the literature of between 232 and 298 SL kg^{-1} VS (Haag et al., 2015; Mast et al., 2014; Ustak & Munoz, 2018).

Based on the yields and N_2O emissions reported by Van Groenigen et al. (2004), energy yield-scaled emissions of fertilized maize on clay soils can be estimated to be between 55.4 and 183.0 g $CO_{2\text{-eq}}$ kWh^{-1} (calculation based on Doehler et al., 2013). On sandy soils, however, energy-scaled emissions only range between 4.0 and 41.5 g $CO_{2\text{-eq}}$ kWh^{-1}, which could be attributed to higher yields and lower emissions from coarser soil, and are in the range of cup plant (20.6 ± 5.6 g $CO_{2\text{-eq}}$ kWh^{-1}) in this study. Therefore, yield potential and the GHG emissions at the respective site are decisive for the opportunity cost of replacing maize with cup plant.

Replacing maize with cup plant in fairly fertile low mountain ranges in central Europe would reduce the energy yield per hectare by 30%. If temperate grassland has to be converted into cropland due to the 30% lower area productivity of cup plant, 1661–2800 kg $CO_{2\text{-eq}}$ more would be emitted because of carbon stock changes (over 20–30 years) due to land-use change (Kindred et al., 2008; Poeplau et al., 2011; Smith et al., 2010). This should be taken into consideration in evaluations of the GHG mitigation of a less productive biomass crop (Searchinger et al., 2009). Taking into account indirect emission due to changes in land use, cup plant would have around 10%–70% higher GHG emissions than maize. Compared with areas that are better suited for maize cultivation, emissions from land-use changes by replacing maize with cup plant would be relatively low at the low mountain ranges sites tested. Moreover, the numerous ecosystem services provided by cup plant at these marginal sites, for example, soil protection, benefits for biodiversity, and C sequestration (Emmerling, 2014; Grunwald et al., 2021; Grunwald et al., 2020; Schorpp & Schrader, 2016), must be taken into account in this assessment (Searchinger et al., 2018).

5 | CONCLUSIONS

Measuring GHG emissions over 2 years from cup plant and maize fields on marginally productive low mountain range sites, which are susceptible to high N_2O emissions because of their soil water regime, showed that cup plant fields emitted substantially less N_2O than the corresponding maize fields. Higher NO_3^- contents in maize fields throughout the year appeared to increase the risk of N_2O emissions, although NO_3^- was not the single most important driver of this. However, the apparent N recovery and productivity of cup plant in this study were low, suggesting a significant unproductive loss of reactive N. It is unclear whether this unaccounted-for N was lost to the environment, where it potentially caused negative external effects, or was bound in belowground biomass. This high unaccounted-for N might be related to the high N rates applied, indicating that an optimized N management in cup plant is needed in order to improve its efficiency and sustainability.

The substantially lower soil-borne N_2O emissions from cup plant were in contrast to the lower area productivity, which needs to be compensated with indirect land-use change and corresponding emissions. However, in central European low mountain ranges, the potential land-use change-related emissions on top of the direct GHG emissions of cup plant are relatively low, because the yield difference between cup plant and other biomass or food crops is low compared with higher yielding areas. When land-use change emissions in relation to cup plant cropping are included, emissions must also be added to the annual system due to the higher input, for example, synthetic fertilizer production, herbicide production/use, and fuel consumption for tillage operations. Furthermore, in contrast to annual crops, cup plant offers a variety of ecosystem services, that is, it mitigates soil erosion, which makes this crop particularly suitable for sloping sites in low mountain ranges. It is difficult to take these ecosystem services into consideration in the sustainability assessment, since they cannot easily be expressed as GHG emissions, but they might offset GHG savings with regard to overall sustainability of biomass production. On sites where overall GHG emissions (direct and from indirect land-use change) were not substantially higher than those from maize, cup plant is the preferable option due to the multiple benefits of this perennial crop on low mountain range sites with challenging water regimes.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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