Title

Dry-Seeding Rice Increases N Losses but Reduces Global Warming Potential Compared to Water-Seeding

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Dry-Seeding Rice Increases N Losses but Reduces Global Warming Potential Compared to Water-Seeding

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Abstract

Flooded rice systems are critical for global food security but contribute significantly to anthropogenic greenhouse gas (GHG) emissions due to high methane (CH_4) production in anerobic soils. Herbicide use in conventional rice systems has also created selection pressure for herbicide resistant aquatic weed species that threaten yields. Dry seeding (DS) rice, which in California includes early season drainage events, has been shown to reduce CH_4 emissions and shift weed species emergence for improved control compared to continuously flooded water-seeded systems (WS). The effect of these drainage events on nitrogen (N) fertilizer losses and nitrous oxide (N_2O) emissions, however, are not well understood. In a two-year study we quantified the effects of early season drainage events utilized in DS rice on global warming potential (GWP) (CH_4 + N_2O in CO_2 eq.), nitrate (NO_3^-) accumulation, and N fertilizer losses as measured by the difference in crop N-uptake compared to a WS control. Despite 1.06 kg ha^{-1} more N_2O emissions in the DS system the GWP was 4,610 CO_2 eq. kg ha^{-1}, a 42% reduction compared to 7,983 CO_2 eq. kg ha^{-1} in the WS system. This was due to a 46% reduction in CH_4 in the DS (126 CH_4 kg ha^{-1}) relative to the WS (235 CH_4 kg ha^{-1}) system. Nitrate accumulation in the DS system amounted to 25.9 kg N ha^{-1}, and subsequent N losses via denitrification likely contributed to the 22.4 kg N ha^{-1} less crop N-uptake in the DS system. These results suggest that DS rice has potential for improved environmental impact via GWP reductions. Future research should consider the effects of increased pre-plant N application rates and timing for improved N management, a quantification of annual GWP including CO_2 emissions, and changes in soil organic carbon stocks in DS rice.

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1. Introduction

Rice (*Oryza sativa* L.) is a staple crop for roughly 3.5 billion people, with global yield trend projections suggesting a 42% increase between 2005-2050, which is short of estimates that crop production needs to double to meet growing demand (Ray et al., 2013). Though these and similar estimates are grounded in neoliberal market assumptions that these rates of increase are inevitable (Holt-Giménez & Altieri, 2012; Tilman et al., 2011), rice as a direct human consumption crop with significant cultural importance in Asia has critical food security implications (Seck et al., 2012). Appropriate production increases should be realized through increased yields, as agricultural land expansion would be ecologically catastrophic with 38% of earth's terrestrial surface currently in cultivation (Foley et al., 2011). Agriculture intensification is however associated with environmental concerns (Tilman, 1999); thus, rice yields need to be maintained or increased while limiting negative environmental impacts (Godfray & Garnett, 2014; Yuan et al., 2021). Rice currently accounts for 48% (Carlson et al., 2017) of cropland greenhouse gas (GHG) emissions or 1.5% of total anthropogenic emissions globally. Therefore, current and or increased yields need to be met with reductions in the crop's global warming potential (GWP).

In California, rice production occurs on approximately 200,000 ha in the Sacramento Valley with yields among the highest in the world, averaging 9.6 t ha^{-1} over the last decade (USDA, 2010-2019). Given that these yields are roughly 75% of the state's yield potential (Espe et al., 2016; Yuan et al., 2021), further yield gap closure is likely to be difficult due to decreased return on additional inputs, and the degree of sophistication in management required to accommodate spatial and temporal variation in weather, soil properties, pest pressure, etc. (Lobell et al., 2009). Rice systems in CA have a relatively low yield-scaled GWP due to high yields and moderate GWP (Yuan et al., 2021). However, there is still potential for GWP reductions through alternative water management practices (Adviento-Borbe et al., 2013; Balaine et al., 2019; LaHue et al., 2016; Pittelkow et al., 2014). Thus, the goal for California rice production should be to at least maintain yields while reducing GWP.

High GWP in rice systems is attributed predominantly to CH_4 emissions which make up 11% of all anthropogenic CH_4 emissions (Smith et al., 2014; Linquist et al., 2012) . In California, CH_4 emissions in rice production can be high for several reasons. First, rice is established using a water seeded (WS) system (Linquist et al., 2018), which involves aerially sowing seeds into flooded fields with the flood maintained until roughly three weeks before harvest when fields are drained. Second, due to restrictions on burning crop residues, rice straw is typically incorporated with fields flooded during the winter fallow to encourage decomposition (Hill et al., 2006; Linquist et al., 2006). This practice, while providing a partially recreated historical flyway for migratory waterfowl (Hill et al., 2006), also increases CH_4 emissions during the winter fallow period and subsequent growing season (Bossio et al., 1999; Fitzgerald et al., 2000; Linquist et al., 2018). Methane is the product of decomposition under anaerobic conditions with low redox potential (Conrad, 2007), thus prolonged flooding with high soil carbon inputs via straw incorporation contribute to high annual CH_4 emissions in California rice systems (Linquist et al., 2018). Practices such as alternate wetting and drying (AWD) have shown to be effective at reducing CH_4 emissions in these systems, while maintaining comparable yield (Balaine et al., 2019; LaHue et al., 2016; Perry et al., 2022) but have not been adopted in California possibly due to a lack of yield or other benefit to growers.

A major challenge for sustaining California rice systems' production efficiencies, that should provide motivation for the consideration of alternative crop management, is the prevalence of herbicide resistant weeds. The heavy clay soils that rice is grown on in California are either not well suited to other crops or are perceived so and are thus continually cropped to rice in most cases (Hill et al., 2006; Rosenberg et al., 2022). The lack of crop rotation and reliance on herbicides with similar modes of action has resulted in California rice systems having some of the most prevalent herbicide resistance in the world, with 11 of the 15 herbicides labeled for use in the state having confirmed resistance to at least one weed species (Driver et al., 2020; Hanson et al., 2014). This has been identified as the largest threat to conventional California rice production (Brim-DeForest et al., 2017; Pittelkow et al., 2012). In the absence of other management approaches to mitigate weed pressure, growers often try to control weeds with increased herbicide applications or combinations of herbicides but this has increasingly limited effect on controlling weed populations (Iwakami et al., 2019; Valverde, 2014).

Dry-seeding (DS) rice is an alternative crop establishment practice that is not widely utilized in California, possibly due to the added complexity during the crop establishment period, but which has promise for controlling herbicide resistant weeds when practiced in rotation with WS (Hill et al., 2006; Linquist et al., 2011) and reducing CH_4 emissions (LaHue et al., 2016; Linquist et al., 2018; Pittelkow et al., 2014). Where DS is practiced in California, the majority of N fertilizer is typically applied as aqua-ammonia injected in bands 7-10 cm below the soil surface (similar to WS). Seed is then drilled into the seed bed or broadcast on the surface and lightly harrowed in. Due to lack of spring precipitation, fields are flush irrigated and drained two to three times to establish the rice crop. At the end of the establishment period (roughly one month after seeding) the field is flooded with a “permanent” flood that is maintained until the harvest drain. This system recruits different weed species and allows for the use of herbicides with different modes of action than those common in WS systems, allowing for improved control of herbicide resistant weeds and similar yield potential to WS rice (Brim-DeForest et al., 2017; Linquist et al., 2011; Pittelkow et al., 2012).

In terms of GWP, early season drains in California DS rice introduce the possibility for increased N_2O but reduced CH_4 emissions (Burger & Horwath, 2012; Lagomarsino et al., 2016; Pittelkow et al., 2014). In WS systems, N_2O emissions are often negligible as the field are flooded for the entire season (Adviento-Borbe & Linquist, 2016; Simmonds et al., 2015). Flush-drain events at the onset of the season to establish the crop create fluctuating aerobic and anaerobic soil conditions that are prone nitrification-denitrification N losses, both of which can result in N_2O emissions (Buresh et al., 2008; Burger & Horwath, 2012). Studies that have quantified N_2O emissions from DS systems in California have reported N_2O emissions during this stage (Adviento-Borbe et al., 2013; Burger & Horwath, 2012; Peyron et al., 2016). Methane reductions in DS relative to WS systems are directly related to a reduced period of soil submergence and anoxic conditions during early season drains when C mineralization is likely occurring with subsequent carbon dioxide (CO_2) emissions via microbial respiration (Conrad, 2007; Ko & Kang, 2000). At question is if a potential increase in N_2O offsets the reduction in CH_4 emissions with regard to GWP in DS systems. The effect of DS drainage events versus WS management on GHG emissions has been examined in only a few of studies which compared the two systems using a stale seedbed practice (Burger & Horwath, 2012; Pittelkow et al., 2014), and another that compared both systems with an additional AWD practice (LaHue et al., 2016), where the DS flush-drain events lead to increased N_2O emissions but reduced CH_4 led to significant GWP reductions. However, only a small portion of the total seasonal N application rate was applied preplant in these studies, with the rest applied just before the permanent flood.

As mentioned, many DS California growers apply all or most of the seasonal N rate pre-plant, which may lead to higher N_2O emissions during the early season irrigation flushes.

The fate of preplant N fertilizer in DS systems that utilize irrigation flushes, where the majority of the N rate is applied preplant, is thus not well understood. In WS systems fertilizer N is well conserved due to sub-surface placement in oxygen depleted soils (Buresh et al., 2008; Linquist et al., 2009, 2011). Direct seeded rice (which include both WS and DS systems) are gaining increased interest around the world, especially where rice has been historically hand-transplanted and labor availability is now limited (Farooq et al., 2011). Dry seeded systems outside of California typically surface apply N in splits or pre-permanent flood to mitigate losses and improve N recovery efficiency (NRE) (Biloni & Bocchi, 2003; Mahajan et al., 2011; Richmond et al., 2018). Surface applied N is associated with a number of losses including ammonia volatilization and leaching, both of which have been found to be minor loss pathways in the California context (Chuong et al., 2020; Liang et al., 2014).

Objectives

Given the above environmental concerns and existing knowledge gaps, a two-year study was conducted comparing a DS and WS system with the following objectives. First, to examine early season N dynamics (particularly nitrification) and quantify crop N losses. We hypothesized that the DS flush-drain cycles will lead to increased nitrification of N fertilizer, which is likely lost via denitrification (or other pathways), thus resulting in a significant reduction in plant N-uptake in the DS treatment compared to the WS control. Second, to quantify N_2O and CH_4 emissions and the GWP of each system. We hypothesized that N_2O emissions would increase during the flush-drain cycles in the DS treatment but that CH_4 reductions would result in a reduced GWP compared to the WS control.

2. Methods and Materials

2.1. Site details

Field experiments were conducted at the California Rice Experiment Station near Biggs, California (39°27’ 31” N,121 44’ 18”) in 0.22-0.25-ha fields during the 2019 and 2020 growing seasons. Historical cultivation and management practices on these fields follows the typical WS rice systems in California, with harvest straw incorporation followed by winter fallow season flooding to encourage decomposition (Linquist et al., 2006). The 2019 and 2020 experiments were conducted in the same general area but in separate fields. Soils at the station are an Esquon-Neerdobe complex, classified as fine smectic, thermin Xeric Epiaquers and Duraquerts. Following land preparation in each spring and prior to fertilization, five soil samples were collected from each field from the plow layer (0-15 cm) and were composited for background soil analyses. After air drying, samples were ground to 2mm and sieved before being sent for texture analysis at the UC Davis Analytical Laboratory using a hydrometer (Sheldrick & Wang, 1993). The 2019 field was found to have 25% sand, 29% silt, and 46% clay, and the 2020 field had 27% sand, 28% silt, and 45% clay. Additional samples were sent to the Midwest Laboratories (https://midwestlabs.com) for analysis of other background soil characteristics. The 2019 field had a soil pH of 5.4, CEC of 32.7 $cmol_c\ kg^{-1}$, and 2.4% organic matter. The 2020 field had a soil pH of 5.1, CEC of 36.9 $cmol_c\ kg^{-1}$, and 3.2% organic matter.

2.2. Experimental Design, Treatments and Management

In 2019, the experiment was established as a randomized complete block design with two water management treatments (DS and WS) and three replicates for each treatment. The field was fertilized preplant with aqua-ammonia at a rate of 168 kg N ha^{-1} injected to a depth of 7-10 cm in bands spaced 22 cm apart which is consistent with grower N applications in California. In

2020, a split-plot design was used with N treatments of 168 kg N ha^{-1} and 0 kg N ha^{-1} as main-plots and water treatments as sub-plots (DS +N, DS -N, WS +N, WS -N). To ensure phosphorus and potassium were non-limiting, fields were fertilized with triple super phosphate at 42 and 59 kg P_5O_2 ha $^{-1}$ in 2019 and 2020, respectively: and muriate of potash at 56 and 65 kg K_2O ha $^{-1}$, respectively. In both years the Calrose medium grain variety M-206, a typical variety grown by California rice growers, was drill seeded at a rate of 112 kg seed ha^{-1} at a 2 cm depth.

In both years the fields were managed as a DS field. Within each field, the WS treatments were established by inserting 0.44m^2 PVC rings down to the hard pan (20 cm) to create a water seal and avoid lateral flow. During the early part of the season (before permanent flood in the DS) when the DS field was being flushed to establish the rice, an irrigation system was established to ensure the WS treatment rings remained flooded. The irrigation system consisted of 3/4" polyethylene tubing suppling water to each WS ring and KerickValve™ PVC mini float valves (1.5 gpm at 60 psi) to maintain the floodwater depth at 10 -15 cm. To mitigate reduced seed emergence from seed buried below the soil surface in the WS treatments, pre-soaked (24hr prior) seed was broadcast onto the soil surface in WS rings at a rate of 100 kg seed ha^{-1}. This seeding rate is lower than the recommendations for California rice and was intended to supplement reduced emergence from the drilled seed. In order to make sure the "ring" effect was consistent between the WS and DS treatments, similar rings were also established for the DS treatment. Holes were drilled into the sides of the DS rings to ensure the soil inside the rings dried at a rate similar to the surrounding field. To avoid potential effects of sampling on parameters measured, separate rings were established for each unique sampling category (soil sampling, GHGs, N-uptake, and yield) for both DS and WS. The main fields were flush irrigated for 24-48hrs one day after seeding (DAS) on June 14 in 2019 and 3 DAS on June 1 in 2020

(NOTE: these dates were when the WS treatments were also initiated and remained flooded for the rest of the season). The flush-drain cycle occurred twice each year before fields were permanently flooded on July 3 in 2019 and July 21 in 2020 until three weeks prior to harvest. Soil volumetric water content was measured throughout the experiment using Em50® loggers (ECH$_2$O System, Meter Group). Due to poor stand establishment in some DS rings in 2020, additional rings were added prior to the permanent flood for GHG sampling, N-uptake, and yield.

2.3. Extractable Nitrogen

During the 2019 growing season, two soil cores were collected from inside designated DS +N and WS +N soil sampling rings every 2-3 days during the flush-drain cycles. Different rings were sampled during each dry down period in each year to avoid over sampling the relatively small area. In 2020, soil cores were only collected from rings at the beginning and end of each dry down event to account for total NO_3^- accumulation during each dry down; however, outside of the rings in the DS +N and DS -N main plots, soils were sampled every 1-3 days to quantify the rate of nitrification. Cores were collected with a 3.5cm diameter Dutch auger at 0-15 cm depth, and at 15 cm spacing laterally across the width of the field, perpendicular to the aqua ammonia bands. Samples from the same ring or main plot were composited, homogenized, and stored on ice for < 24h before they were extracted in triplicate subsamples with 2M KCl in a 1:10 soil to solution ratio. Sub-samples were placed on a mechanical shaker for one hour before being filtered to a clear extract through Whatman No. 42 filter paper (GE Healthcare UK, Limited, Buckinghamshire, UK). Extracts were analyzed colorimetrically for NH_4-N (Forster, 1995) and NO_3-N (Doane & Horwáth, 2003). Gravimetric soil moisture data was collected by taking 50-75 g of each sample and placing them in an oven dryer (105°C) until constant weight was achieved.

Bulk density soil cores were collected from each block, during each soil sampling event to a depth of 0-15 cm and were used to determine extractable N in units of kg N ha^{-1}.

2.4. Plant Sampling

At physiological maturity, plants were sampled to determine N-uptake and yield from designated yield rings for all treatments in 2019 and 2020. These rings had no other sampling nor any other manipulation of the area inside each ring during the growing season. To sample, all plants in the ring were cut at soil level, oven dried to a constant dry weight at 60°C, and then separated into straw and grains fractions to obtain grain and straw weights. Grain and straw samples were then ground to a powder and analyzed for total N by combustion at the UC Davis stable isotope facility. Grain yields were adjusted and reported at 14% grain moisture which represents the industry standard. Crop N-uptake was determined as the combined total N content in grain and straw. Nitrogen recovery efficiency for 2020 was calculated using the equation below.

$$NRE = \left(\frac{Fertilized\ N\ Uptake - Unfertilized\ N\ Uptake}{N\ Applied}\right) \times 100$$

2.5. Greenhouse Gas Emissions

Nitrous oxide and CH_4 fluxes where measured every 1-3 days during the flush-drain cycle, and corresponded with soil sampling events, using the static vented chamber method (Hutchinson & Livingston, 1993). After the DS treatments went into a permanent flood, sampling occurred weekly for the remainder of the season. Chambers were established inside designated rings, with the insertion of 29.5 cm diameter PVC collar bases with two 11 cm holes drilled into opposite sides to allow water to flow freely through the chamber in the WS treatment and when fields were flooded for both treatments. Holes were plugged with rubber stoppers in the DS treatments when fields were drained to ensure airtight sealing of the chambers. Collars were inserted at the start of the initial flush at a 15 cm depth and left inside the rings for the remainder of the growing season. Chamber extensions were used during sampling, ranging from 15.3 cm – 120.9 cm in height depending on the height of the crop. Chamber lids which were 7.5 cm in height were lined on the outer surfaces with reflective sheets. The lids sealed the chamber and were equipped with a battery powered mechanized fan (used 1-minute before each sample extraction to homogenize headspace air), a vent tube to equalize pressure inside and outside the chamber, a thermocouple wire to measure chamber air temperature during sampling, and a silicon port where gas samples were collected.

Previous drill-seeded studies in California that assessed GHG emissions recorded diurnal gas measurements during the growing season and observed no significant variation in gas fluxes (Adviento-Borbe et al., 2013; Pittelkow et al., 2014). Thus sampling occurred between 0900-1200h, during which soil temperatures are expected to represent the daily average (Bossio et al., 1999). To sample, 25 mL samples of headspace air were collected in 35 mL syringes at 0, 21, 42, and 63 min and immediately transferred into evacuated 12 mL glass vials with a rubber septa (Labco Ltd., Buckinghamshire, UK) and an added silicon layer to further minimize gas leaks. In 2020, with the establishment of new gas rings to compensate for those with poor germination in the initial set, samples were collected in both the rings we retired and the new rings during two successive sampling events. Gas measurements were the same in both sets of rings and we assumed there was no difference moving forward with the new rings for the remainder of the season.

Gas samples were analyzed for N_2O and CH_4 peak area on a Shimadzu 2014 gas chromatograph (GC) equipped with a ^{63}Ni electron capture detector set at 325°C for N_2O detection and a flame ionization detector (FID) for CH_4. The gas species were separated by a stainless-steel column, with Hayesep D, 80/100 mesh at 75°C. The detection limits for the GC were 0.3 pg s^{-1} N_2O and 2.2 pg s^{-1} CH_4 (Adviento-Borbe et al., 2013). Standards for N_2O and CH_4 were used to calibrate the GC with 95% certified accuracy (Airgas Inc.). Results from the GC were accepted if the standard gas calibrations produced a linear relationship between voltage output and gas concentrations with an $r^2 > 0.99$. Gas fluxes were estimated from peak area for each sample based on the linear relationship and increase of gas concentration over time. Concentrations were converted to elemental mass per unit area (g ha^{-1} d^{-1}) using the Ideal Gas Law, accounting for the chamber volume, the temperature measured during each sample collection, and an atmospheric pressure of 0.101 MPa. Fluxes were computed as:

$$F = \frac{\Delta C}{\Delta t} \times \frac{V}{A} \times \propto$$

where F is the gas flux rate (g N_2O-N ha^{-1} d^{-1}, g CH_4-C ha^{-1} d^{-1}), $\Delta C/\Delta t$ denotes the increase or decrease of concentration in the chamber (gL^{-1} d^{-1}), V is the chamber volume, A is the enclosed surface area (ha), and $\propto$ is a conversion coefficient for elemental N and C (28/44 for N2O; 12/16 for CH_4) (Adviento-Borbe et al., 2013) a conversion coefficient for elemental N and C (28/44 for N_2O and; 12/16 for CH_4). Gas fluxes where the r^2 was < 0.85 but passed the detection tests were not included in the analysis, and fluxes that failed detection were included as zeroes. To determine growing season cumulative emissions, individual flux values were integrated across all time points with a linear interpolation. A detailed description of these analyses can be found in Adviento-Borbe et al. (2013). In 2020, the first two sampling dates (before and during the first irrigation flush, 1 and 3 days after seeding) were only collected from the DS +N and DS -N treatments as there were no differences in water treatment prior to the first drain (Fig 3.). The first spike for the DS +N were added to the both the DS +N and WS +N cumulative emissions, and the same was done for the DS -N and WS -N spike.

3. Data Analysis

Statistical analysis of the data was performed in R-Studio (version 4.1.0 R Core Team, 2021). All data were analyzed as a completely randomized block design, and an analysis of variance was performed using linear mixed effects models and the restricted likelihood method in the lme4 package (Bates et al., 2015). For all two-year models, water treatment (DS +N and WS +N) was treated as a fixed effect with block and year as random effects. Significant differences between treatments were detected using the Tukey pairwise comparison ($P < 0.05$) in the multcomp package in R (Hothorn et al., 2008). Data met assumptions of normality using the Shapiro-Wilk test and homogeneity using visual diagnostic plots. GWP was calculated using the 100-year time scale using radiative forcing potentials with the climate-carbon feedback relative to CO_2 of 34 for CH_4 and 298 for N_2O (Myhre et al., 2013). Separate models for each year were created to analyze CH_4 and GWP data due to a significant year by treatment effect. Data for NO_3-N and N_2O for the WS treatments resulted in almost entirely zero values in both years. In an effort to analyze this data we replaced the zeroes by applying half the detection limit of the method for NO_3-N analyses of 0.01μg N mL^{-1} (Doane & Horwáth, 2003) and the gas chromatographer of 0.3pg s^{-1} for N_2O. We then power transformed the data after adding the minimal values but did not achieve a normal distribution. The reported values are untransformed, as the transformation would have decreased averages for the upper end of the values (DS +N treatment), without gaining increased insight on the predictions from the transformed models. The lack of normality in the N_2O data ($p= 0.018$) led to a lack of normal distribution in the GWP data ($p=0.044$), which we left untransformed as a result of this issue with the N_2O distribution.

For the purposes of the discussion, the "start" of each dry down was when the volumetric water content dropped below 50% (saturation level for this soil) after field drains were initiated.

Observationally, this was when there was almost no puddling on the soil surface across the field. We used this starting point for each dry down period to create a regression analysis of the rate of nitrification in the DS +N treatment using data from all samples collected from rings in 2019 and 2020 and the main-plot DS +N sampling from 2020. Ring and field NO_3^- values from the DS +N plots were the statistically the same in 2020 for the purposes of the linear regression.

4. Results

4.1. Crop N-Uptake and Yields

At harvest in the treatments receiving N fertilizer, crop N-uptake in the WS +N treatment averaged 163 kg N ha^{-1} compared to 140 kg N ha^{-1} in the DS +N treatment across both years (Table 1). In 2019 and 2020 the WS +N treatment had a 26.7 kg N ha^{-1} and 18.0 kg N ha^{-1} (average 22.4 kg N ha^{-1}) higher N-uptake than the DS +N treatment. In the -N treatment (2020 only), N-uptake in the DS -N and WS -N treatments were similar, being 84.4 and 86.2 kg N ha^{-1}, respectively. Based on this, the NRE for the fertilized treatments in 2020 was 47% in the DS +N and 57% in the WS +N. Yields were similar across both years but were different between the two +N water treatments, averaging 11.2 Mg ha^{-1} in the WS +N treatment and 10.1 Mg ha^{-1} in the DS +N treatment. In the treatments that did not receive N, yields were similar between the two treatments and averaged 7.3 Mg ha^{-1} in WS -N and 7.6 in DS -N.

4.2. Early Season NO_3^- Dynamics

In the WS +N treatment, which remained flooded during the early season, NO_3-N values were at or near zero for the duration of the early season in both years (Fig. 1). In contrast, in the DS +N treatment, soil NO_3^- began accumulating 3-5 days after drains were initiated in both years, at the inflection point of < 50% volumetric water content when the fields begin to dry down. The duration of the two dry down periods was 5 and 7 days in 2019 and 7 and 7 days in

2020. Nitrate accumulated during the dry down periods declined to zero or near zero when soils were flush irrigated, or when the permanent flood was initiated. In the +N plots, more NO_3^- accumulated during the second dry down in both years than during the first dry down. Based on the linear regression between the number of dry down days and NO_3-N accumulation, NO_3^- accumulated at a rate of 2.17 kg NO_3-N ha^{-1} day^{-1} (r^2=0.84) (Fig. 2).

Total NO_3-N accumulation was calculated as the sum of the two peaks of each dry down in each year, which corresponded with the last sampling date for each dry down in the DS treatments. Total NO_3-N for DS +N ranged from 20.3 kg NO_3-N ha^{-1} in 2019 to 31.5 kg NO_3-N ha^{-1} in 2020 which was likely related to the increased dry down days in 2020 (Table 1). The average across both years for the DS +N treatment of 25.9 kg NO_3-N ha^{-1} and was significantly higher than the WS +N treatment where no NO_3-N was detected in either year. In 2020, NO_3^- dynamics were also quantified in -N plots to provide insights into residual mineral N pool dynamics. Accumulation in the DS -N treatment followed a similar pattern to the DS +N treatment. In DS -N, nitrate accumulation was greater during the first dry down period (6.3 kg NO_3-N ha^{-1}) than during the second (1.2 kg NO_3-N ha^{-1}) (Fig. 1) and totaled 7.5 kg NO_3-N ha^{-1} (Table 1). Thus, in 2020 fertilizer induced NO_3^- accumulation amounted to 24.0 kg NO_3-N ha^{-1}.

While NH_4^+ was quantified, results were highly variable (data not shown) due to the preplant fertilizer being applied in bands. It was not possible to determine if soil samples were in or between N bands, as they were visually indistinguishable after application.

Table 1

Total extractable NO_3-N accumulation during early-season dry-seeded (DS) dry down, harvest plant N-uptake (grain + straw total N), and yield at N rates of 168 kg N ha^{-1} (+N) and 0 kg N ha-1 (-N). NO_3-N accumulation is the sum of the two dry down peaks in each year, which occurred on the final day of each dry down.

Treatment[a]	Nitrate Accumulation (NO_3-N ha^{-1})			Harvest N-Uptake (kg N ha^{-1})			Yield (Mg ha^{-1})		
	2019	2020	Mean	2019	2020	Mean	2019	2020	Mean
DS +N	20.3	31.5	25.9*a*	117.1	163.4	140.2*a*	9.9	10.2	10.1*a*
	(2.0)	(3.0)	(3.0)	(6.4)	(1.2)	(10.3)	(0.1)	(0.5)	(0.2)
DS -N		7.5			84.4			7.6	
		(2.9)			(3.2)			(0.5)	
WS +N	0.0	0.0	0.00*b*	143.8	181.4	162.6*b*	11.2	11.2	11.2*b*
	(0.0)	(0.0)	(0.0)	(8.3)	(7.1)	(9.7)	(0.2)	(0.8)	(0.4)
WS - N		0.0			86.2			7.3	
		(0.0)			(4.3)			(0.4)	

a Treatments are dry-seeded (DS +N, DS -N) and water-seeded (WS +N, WS -N) fertilized with 168 kg N ha^{-1} and 0 kg N ha^{-1}

b SE of the mean indicated in parentheses. Differing letters following two-year averages represent significant differences between treatments ($p < 0.05$)

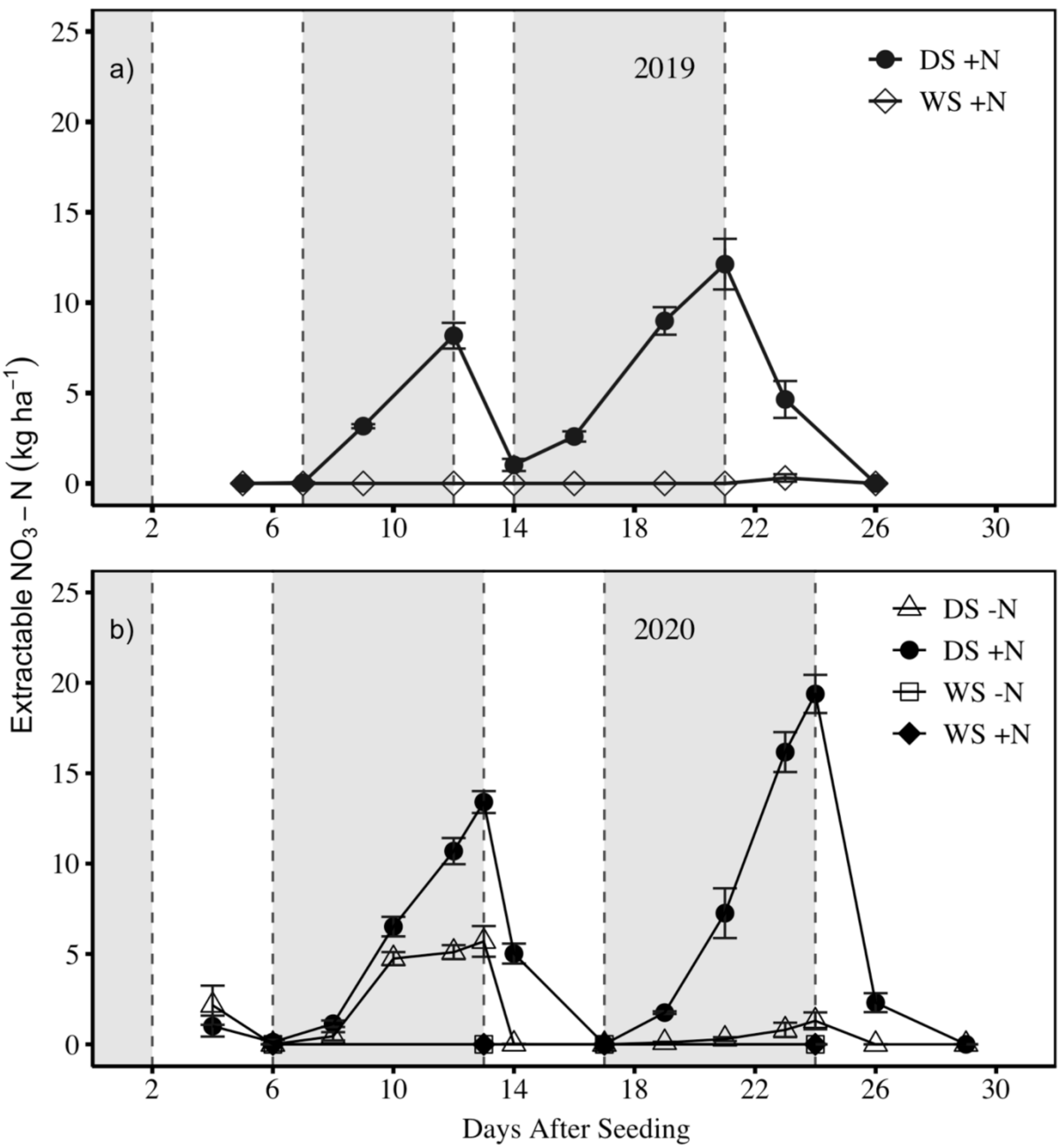

Fig. 1.
Soil extractable NO_3-N during flush-drain cycles: a) 2019 Dry-seeded 168 kg N ha^{-1} (DS +N) and Water-seeded 168 kg N ha^{-1} (WS +N) treatments; b) 2020 DS +N and -N (0 kg N ha^{-1}) and WS +N and -N (0 kg N ha^{-1}) treatments. Grey shading indicates dry down periods for DS treatments, marked by dotted lines to indicate the start and end of the dry down. WS treatments were continuously flooded. Error bars represent the standard error of the mean for each sampling event.

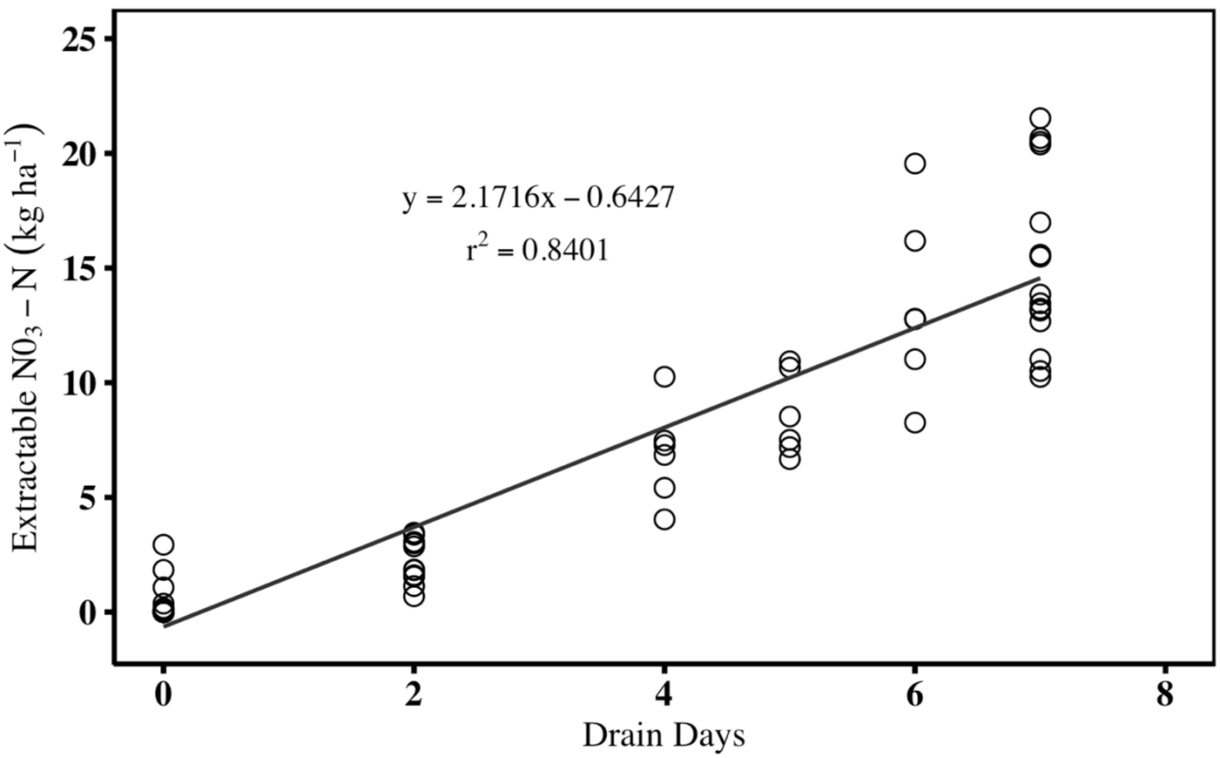

Fig. 2.
Linear regression of soil NO_3-N as a function of the number of dry down days from fully fertilized 168 kg N ha^{-1} dry-seeded (DS +N) treatment. Day zeroes were established when soil volumetric water content dropped below 50%.

4.3. Greenhouse gas emissions

At the onset of sampling in both years, N_2O spikes occurred during the initial flush irrigation event likely due to the denitrification of the existing NO_3^- but were negligible or zero once the drains were initiated and this remained the case for the WS treatments throughout the entire season in both years (Fig. 3). In the DS +N treatment, N_2O fluxes began after the fields were first dried down but with differing flux patterns in each year. In 2019, two N_2O peaks (both above 50 g N_2O ha^{-1} day^{-1}) occurred in DS +N at the end of the second flush and 2 days after the permanent flood. In 2020, emissions spiked over two consecutive sampling dates during the first dry down (both above 150 g N_2O ha^{-1} day^{-1}) after which it dropped to roughly 50 g N_2O ha^{-1} day^{-1}, where it remained until the field went into a permanent flood. Nitrous oxide emissions were at or near zero in the DS +N treatment after the permanent flood (data not shown). Cumulative N_2O emissions in the DS +N treatment were 0.82 kg N_2O ha^{-1} season $^{-1}$ in 2019 and 1.33 kg N_2O ha^{-1} season $^{-1}$. Averaged across both years cumulative N_2O emissions in the DS +N treatment were 1.06 kg N_2O ha^{-1} season $^{-1}$ higher than the WS +N treatment (Table 2). In 2020, the DS -N treatment had an initially high N_2O spike similar to the DS +N treatment but declined rapidly thereafter. Following this initial spike in emissions there were some smaller peaks following the second flush irrigation and during the second dry down period, but N_2O emissions were zero just before the permanent flood was initiated. Cumulative emissions for the DS -N treatment was 0.95 kg N_2O ha^{-1} season $^{-1}$. Thus, fertilizer induced emissions were 0.38 kg N_2O ha^{-1} season $^{-1}$.

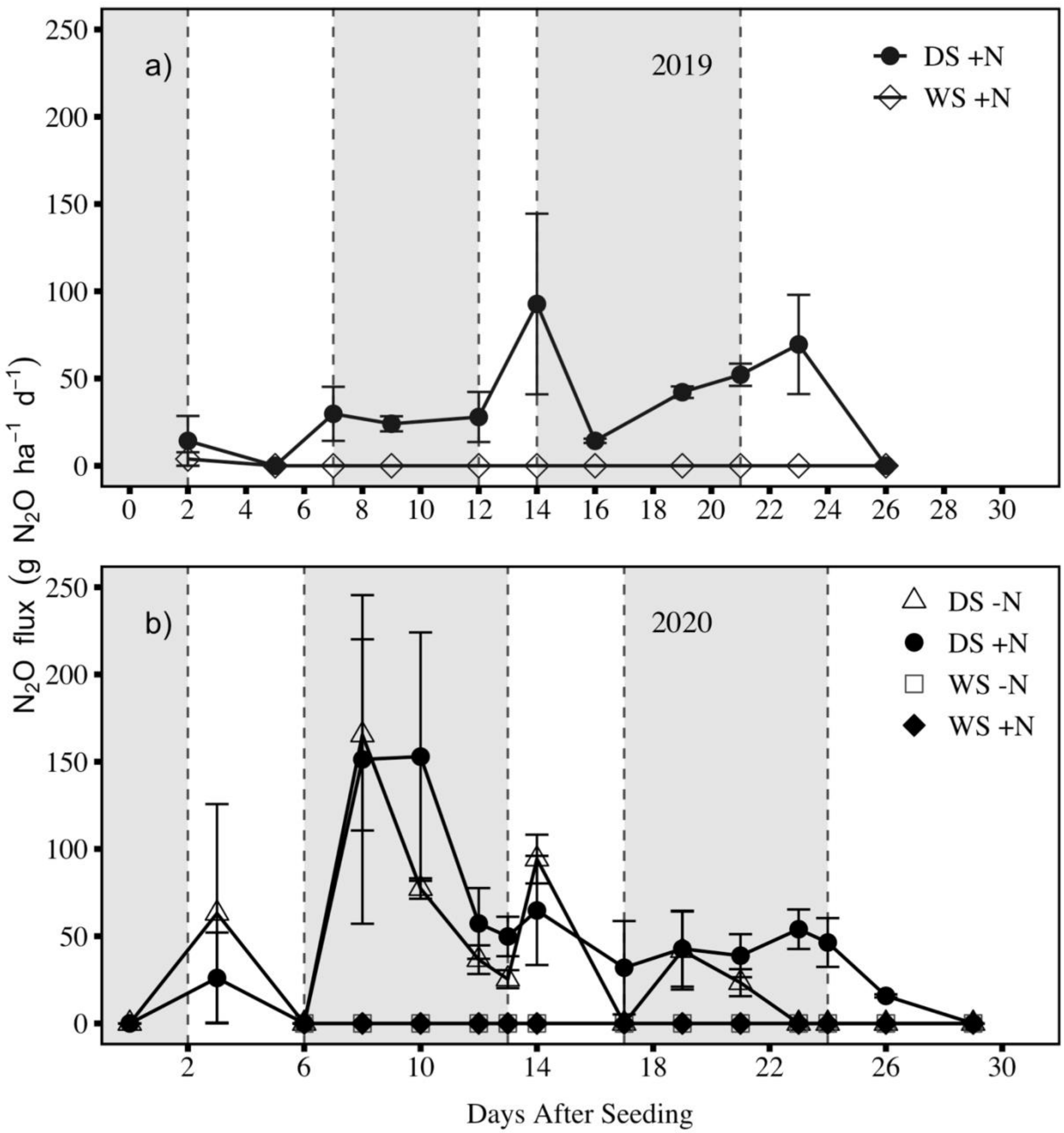

Fig. 3. N_2O fluxes during flush-drain cycles: a) 2019 Dry-seeded 168 kg N ha^{-1} (DS +N) and Water-seeded 168 kg N ha^{-1} (WS +N) treatments; b) 2020 DS +N and -N (0 kg N ha^{-1}) and WS +N and -N (0 kg N ha^{-1}) treatments. Only the first month is included with no detectable N_2O emissions for the remainder of the growing season in both years. In 2020, the first two sampling events were taken from the DS +N and DS -N as there was no water treatment difference before the first drain. Grey shading indicates dry down periods for DS treatments, marked by dotted lines to indicate the start and end of each dry down. WS treatments were continuously flooded. Error bars represent the standard error of the mean for each sampling event.

Table 2
Seasonal N_2O and CH_4 emissions and GWP for the dry (DS) and water seeded (WS) treatments both with fertilizer N (+N) and without (-N). Nitrous oxide emissions occurred only during the flush drain cycles in DS treatments and were not detected the WS treatments.

Treatment[a]	N_2O Emissions (N_2O kg ha^{-1})			CH_4 Emissions (CH_4 kg ha^{-1})			GWP (CO_2 eq. kg ha^{-1})		
	2019	2020	Mean	2019	2020	Mean	2019	2020	Mean
DS +N	0.82 (0.18)	1.33 (0.26)	1.08*a* (0.18)	120.0 *a* (7.9)	132.3*a* (12.0)	126.1*a* (6.97)	4326*a* (242.4)	4894*a* (342.1)	4610*a* (226.57)
DS -N		0.95 (0.16)			146.3 (13.7)			5236 (502)	
WS +N	0.01 (0.01)	0.03 (0.00)	0.02*b* (0.01)	288.2 *b* (30.2)	181.1*a* (22.1)	234.6*b* (29.2)	9802*b* (1024)	6164*a* (751)	7983*b* (991.99)
WS -N		0.06 (0.00)			220.4 (36.9)			7494 (1254)	

[a] Treatments are dry-seeded (DS +N, DS -N) and water-seeded (WS +N, WS -N) fertilized with 168 kg N ha^{-1} and 0 kg N ha^{-1}

[b] CH_4 and GWP were analyzed separately by year for the DS +N and WS +N treatments due to a significant year by treatment effect.

[c] Standard error of the mean indicated in parentheses. Differing letters in each column following averages represent significant differences for DS +N and WS +N ($p < 0.05$).

Methane emissions in the WS +N treatment were detected two weeks after flooding was initiated in each year (Fig. 4). In 2019 in the WS +N treatment, CH_4 emissions increased rapidly from June 28 to July 23. From July 23 to September average daily emissions remained high at 4,880 g CH_4 ha^{-1} day^{-1}. In early September emissions declined rapidly and after the dry down on September 19, emissions were reduced to zero by the second to last sampling date on October 4. In 2020 in WS +N emissions peaked early but then declined through the rest of the season. Furthermore, maximum daily emissions for the WS +N treatment were considerably higher in 2019 compared to 2020 (5,770 vs 3,650 g CH_4 ha^{-1} d^{-1}). Maximum daily emissions in the WS-N treatment in 2020 occurred two weeks earlier (5,470 g CH_4 ha^{-1} d^{-1} CH_4) than WS +N and were the highest of all treatments that year.

In the DS +N treatment there were minor CH_4 fluxes during the first dry down event of each season (230 g CH_4 ha^{-1} day^{-1} in 2019 and 120 g CH_4 ha^{-1} day^{-1} in 2020). This was followed by non-detectable emissions until 1-2 weeks after the permanent flood was initiated at the end of the flush-drain cycles. In 2019, CH_4 emissions had a sustained increase toward a peak in early-September (the early-September date had only one viable sample and thus no standard error). The 2020 flux pattern for DS +N experienced a gradual increase through early-August followed by a decline through to the end of the growing season. Maximum daily emissions for DS +N was 4,700 g CH_4 ha^{-1} d^{-1} in 2019 and 2,740 g CH_4 ha^{-1} d^{-1} in 2020. In the DS-N treatment in 2020 maximum daily emissions (2,780 g CH_4 ha^{-1} d^{-1}) were almost identical to DS +N but occurred two weeks prior, though emissions for both DS treatments followed a very similar pattern all season long. Sampling did not occur intensively during the harvest drain in either season, thus no post drain peaks during this period for any of the treatments were observed. Analyzed across both years, cumulative CH_4 emissions were significantly different between the two +N water treatments, with 235 kg CH_4 ha^{-1} season $^{-1}$ in the WS +N treatment and 126 kg CH_4 ha^{-1} season $^{-1}$ in the DS +N treatment. However, there was treatment by year interaction as the two +N treatments were not significantly different in 2020 with 181 kg CH_4 ha^{-1} season $^{-1}$ and 132 kg CH_4 ha^{-1} season $^{-1}$, in the WS +N and DS + N treatments, respectively (Table 2). The overall trend was still similar, and across both years, the DS +N treatment reduced CH_4 emissions by 46% compared to WS +N. Cumulative CH_4 emissions in the -N treatments in 2020 were the same, despite WS -N having considerably higher average emissions of 220 kg CH_4 ha^{-1} $season^{-1}$ compared to 146 kg CH_4 ha^{-1} $season^{-1}$ in DS -N. The percentage decrease in CH_4 emissions in the -N treatments was 34%.

The trend in GWP between the two +N water treatments was generally similar to the trend in CH_4 because CH_4 represented 94% and 92% of CO_2 eq. emissions in 2019 and 2020 respectively in the DS+N treatments. While N_2O was higher in the DS+N treatments, it had a relatively minor effect on overall GWP. Across both years GWP was significantly reduced by 42% in the DS+N treatment, but similar to CH_4 emissions this was only significant in 2019 due to a treatment by year interaction. Similarly, with respect to the non-fertilized treatments in 2020, there was no significant difference in CH_4 emissions or GWP between the two WS N treatments, nor between the two DS N treatments.

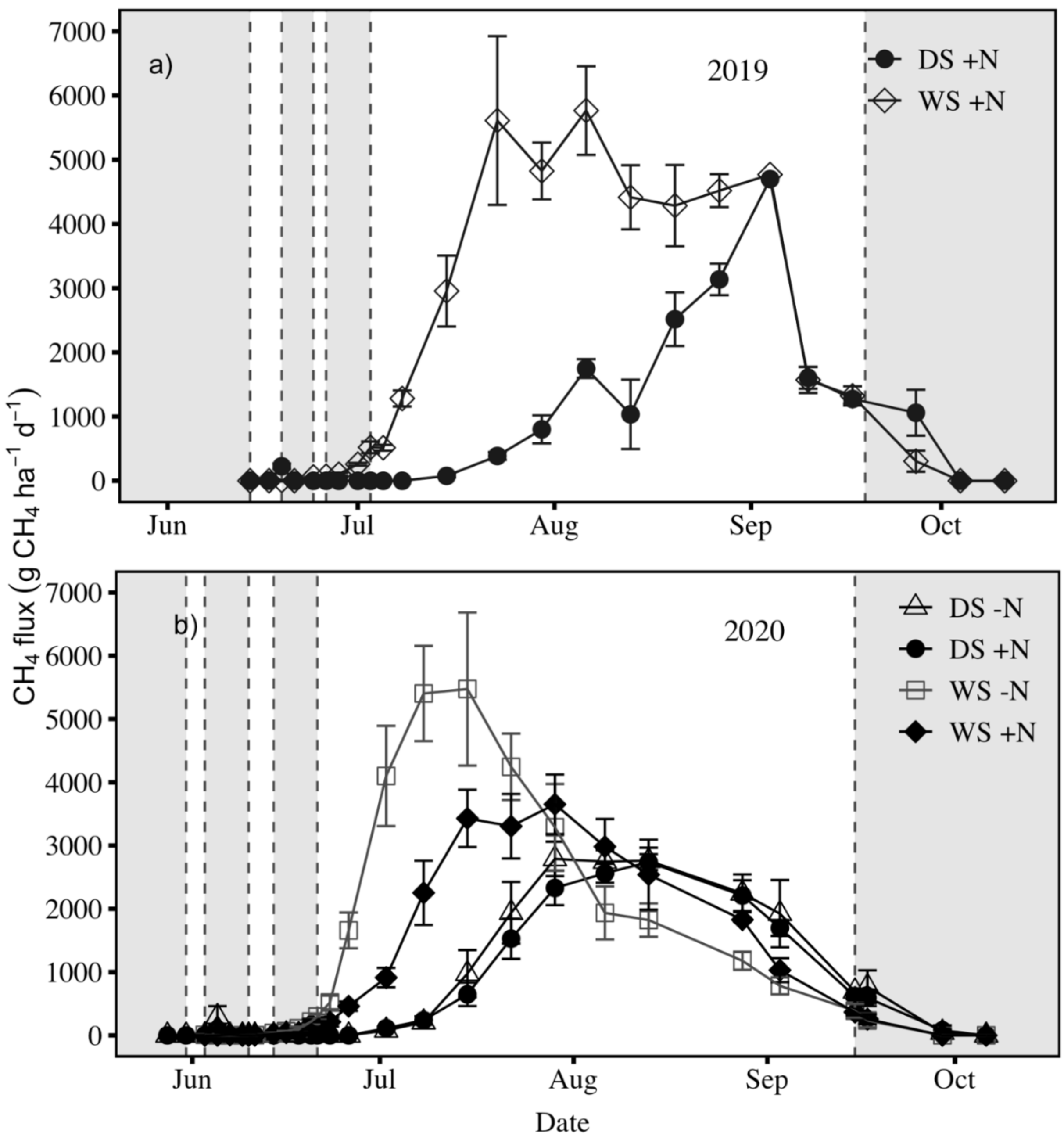

Fig. 4.
Growing season CH_4 fluxes: a) 2019 N fertilized Dry-seeded (DS +N) and Water-seeded (WS +N) treatments; b) 2020 N fertilized and unfertilized DS +N and -N and WS +N and -N treatments. Grey shading indicates dry down periods for DS treatments in the early part of the season and harvest drains for all treatments in late September, marked by dotted lines to indicate the start and end. WS treatments were continuously flooded. Error bars represent the standard error of the mean for each sampling event.

5. Discussion

5.1 Global Warming Potential in DS Rice

Cumulative CH_4 emissions in the WS +N treatment of 235 kg CH_4 ha^{-1} $season^{-1}$ were consistent with the average emissions for water seeded rice in California of 218 kg CH_4 ha^{-1} $season^{-1}$ (Linquist et al., 2018). Sampling was not intensive after the final drain in preparation for harvest and thus the spike in CH_4 emissions, which is attributed to gases being released from the soil, was not observed; however, these end of season emissions can account for about 10% of seasonal emissions (Adviento-Borbe et al., 2015). Furthermore, the cumulative emissions do not include emissions from the winter fallow, which have been shown to average 27% of annual CH_4 emissions in California (Fitzgerald et al., 2000; LaHue et al., 2016; Linquist et al., 2018; Pittelkow et al., 2013). The addition of N fertilizer in the WS +N treatment did not significantly influence CH_4 emissions (Table 2) relative to the WS -N treatment, as reported on by others when comparing the 168 kg N ha^{-1} application rate (Linquist et al., 2012; Pittelkow et al., 2014).

Confirming our hypothesis, CH_4 emissions in the DS+N treatment were significantly reduced by 46% (Table 2). This reduction is similar to the 47% reduction reported by (Pittelkow et al., 2014) when DS and WS treatments were compared in systems employing stale seedbeds and the 66% reduction reported by LaHue et al., (2016) when alternate wetting and drying (AWD) was incorporated into both systems. Since most of the early season CH_4 emissions are attributed to decomposition of residue from the previous season (Conrad, 2007; Islam et al., 2018), maintaining aerobic periods during the early part of the growing season allows the residue to mineralize with CO_2 being emitted instead of CH_4 (Miyata et al., 2000). In continuous rice systems as is common in California where residues are not burned but rather left in the field to decompose over the winter fallow, considerable amounts of residues can remain at the end of the fallow period (Linquist et al., 2006) and resulting CH_4 emissions early in the following cropping season can be high (Chidthaisong & Watanabe, 1997), as seen in the WS treatments in this study (Fig. 4). To mitigate these high early season CH_4 emissions, some studying AWD have reported that drainage earlier in the season may more effectively reduce seasonal CH_4 emissions, compared to later drainage events (Islam et al., 2018; Tariq et al., 2017). The DS system does something similar by maintaining a wet but mostly aerobic state in the early part of the season. Methane emissions in DS systems have been shown to be further reduced using AWD practices (Linquist et al., 2015; Runkle et al., 2019) but that was not the focus of this study.

The effect of DS on GWP was similar to that seen for CH_4 emissions, as CH_4 emissions accounted for 99.9% and 93% of CO_2 eq. kg ha^{-1} season^{-1} in the WS +N and DS +N treatments, respectively (Table 2). Overall, GWP was reduced by an average by 42% in the DS+N treatment. While CH_4 emissions were dominant, N_2O emissions from the DS +N treatment averaged 1.08 kg N_2O ha^{-1}. These emissions were in part due to residual mineral soil N as seen in the DS -N treatment in 2020, but increased, as observed in other studies, due to preplant N fertilizer applications (Burger & Horwath, 2012). The fertilizer-induced N_2O emission factor (FIEF), defined as the emissions of N_2O-N kg ha^{-1} from fertilized plots minus the zero N control, was 0.14% in 2020, which is lower than previously reported FIEF of 0.31% (Akiyama et al., 2005). This suggests that the N_2O emissions in the DS system here, which was implemented in fields with a WS management legacy is dependent largely on residual mineral N which is no doubt highly dependent on large N inputs, is an important consideration. In DS systems in other parts of the world, most of the N fertilizer is applied just before the permanent flood is established, although some starter N may be applied at planting (Dunn et al., 2014; Norman et al., 2009; Richmond et al., 2018). When fertilizer N is applied just before the permanent flood, N_2O

emissions are generally kept low in DS systems (Adviento-Borbe et al., 2013; Simmonds et al., 2015) particularly if mineral N is limited during the flush and drain events.

5.2. Potential Causes of Lower N Uptake in DS

Plant N-uptake in the DS +N treatment averaged 22.4 kg N ha^{-1} less than the WS +N treatment across both years (Table 1). The only difference in management between these two systems was during the establishment phase (first 3 to 4 weeks) where the DS treatment received flush irrigation events to establish the rice crop while the WS treatment remained flooded. Ammonia volatilization can be a major source of N loss in DS systems with surface applications (Buresh et al., 2008; Dillon et al., 2012; Norman et al., 2009), but has been shown to be minimal both in California and elsewhere when N is applied subsurface (Chuong et al., 2020; T. Q. Liu et al., 2015). In this study, nitrification was clearly seen during the drain events with NO_3^- accumulating in the DS treatments compared to almost no NO_3^- detected in the WS treatments (Fig. 1). Importantly, while NO_3^- can be taken up by rice (Duan et al., 2007; Yoneyama et al., 2016), in the system discussed here, the seeds are either just germinating or the plants are too small to take up a significant amount of N (Yoneyama et al., 2016).

Preplant N applications followed by irrigation flushes creates suitable redox potential for nitrification, even with subsurface applications, where accumulated nitrates (NO_3^-) can be lost when the field is reflooded via leaching or denitrification (Buresh et al., 2008; Burger & Horwath, 2012; Linquist et al., 2011; Patrick & Wyatt, 1964). Due to the heavy clay soils in the Sacramento Valley, downward percolation is limited (LaHue & Linquist, 2019), and NO_3^- leaching is minimal (Liang et al., 2014). Thus, most of the early-season N losses in California DS rice are likely via N_2 and N_2O emissions (Buresh et al., 2008; Firestone & Tiedje, 1979). The nitrification-denitrification loss pathway results in N being primarily lost as dinitrogen (N_2) gas with smaller emissions of N_2O (Firestone & Tiedje, 1979; Morley et al., 2014). While N_2 is only produced during denitrification, N_2O can be produced during both nitrification and denitrification (Zhu et al., 2013). In this study N_2O was emitted during periods where both nitrification (drying) and denitrification (after flooding) would be expected (Fig. 3), with seemingly significant contributions from both emissions pathways (i.e., higher emissions during flooded periods in 2019, and higher emissions during the first dry down in 2020). From an N budget standpoint, the amount of N lost via N_2O was minimal and averaged 0.69 kg N_2O-N ha^{-1} across both years. It is thus assumed that most of the rest of the N lost was in the form of N_2.

On average, nitrification led to an accumulation of roughly 25.9 kg NO_3-N ha^{-1} in the DS +N system, of which over 75% was from the applied fertilizer N (based on 2020 data). The amount of NO_3^- accumulated was similar to the amount of N lost based on the difference in plant N uptake between the DS+N and WS+N treatments (22.4 kg N ha^{-1}; Table 1), further suggesting that nitrification and the subsequent loss of NO_3^- is the main N loss pathway difference in these two systems. Given the potential loss of NO_3^- in DS rice, understanding the rate of NO_3^- accumulation during flush drain cycles is critical for optimal N management and environmental stewardship. Nitrate accumulated at a rate of 2.17 kg NO_3-N ha^{-1} day^{-1} in the DS +N treatment (Fig. 2). This rate is similar to the 2.02 kg NO_3-N ha^{-1} day^{-1} Linquist et al. (2011) reported. Their study was conducted over a wide range of fields and soils but did quantify NO_3^- accumulation in WS systems that experienced dry-downs early in the season to allow for foliar active herbicide applications (usually beginning 1-2 weeks after planting) before much of the preplant fertilizer N was taken up by the crop. Importantly, in both this study and that of Linquist et al. (2011) most or all of the N rate was applied as a preplant N application at the recommended rates of 150 to 180 kg N ha^{-1} (Linquist et al., 2009; Williams et al., 2010). Nitrate accumulation rates will vary depending on the fertilizer N application rate (and the amount of mineral N in the soil) as was clear from our DS -N treatment in 2020 that experienced less NO_3^- accumulation (Fig. 1; Table 1). Furthermore, nitrification rates vary depending on temperature (Sabey et al., 1956; Schmidt, 1982), soil organic matter (Sabey et al., 1956; Schmidt, 1982), straw management which may affect N immobilization (Said-Pullicino et al., 2014), pH (Dancer et al., 1973), and soil texture among other factors. These factors may have led to the poorer relationship between NO_3^- accumulation and time in Linquist et al. (2011) study at multiple locations (R^2= 0.41) than in the current study at one location (R^2=0.84).

5.3 Improving the DS System

In the DS system here, GWP was much reduced, suggesting its promise as an effective rice cropping system that reduces environmental impact. While yields were reduced in the DS +N treatment, this reduction was likely due to reduced N uptake due to early season N losses as there was no weed pressure in either treatment. Previous research comparing these systems has indicated that yield potential is similar when appropriate N rates are added (Linquist et al., 2011; Pittelkow et al., 2012). Therefore, to realize this yield potential, N fertilizer needs to be managed differently than in WS systems. This could be achieved with additional preplant N fertilizer, to account for predicted losses. Based on this research it is not clear how much additonal N would be needed. This could be estimated via the amount of NO_3^- accumulation in the DS +N treatment (25.9 kg N ha^{-1} in our case), though this varies based on the dry down duration (Fig. 2). An alternate estimate could be based on the difference in N uptake between the WS and DS system and the NRE of applied fertilizer. In the DS system the NRE was 47% and the N uptake diffrence was 22.4 kg N ha^{-1}. Based on this, an extra 48 kg N ha^{-1} would need to be added. While further research is required in determining the actual replacement N rate, the additional N, if applied preplant may increase early season N_2O emissions, the GWP of the system, and the broader carbon footprint associated with the industrial fixation and transport of the additional inorganic N fertilizer.

The second option, is to apply all of the N fertilizer just before the permanent flood. This and split N applications are typical of DS systems elswhere and have led to high NRE and yields (Dillon et al., 2012; Dunn et al., 2014; H. Liu et al., 2015). Few studies have compared these systems side-by-side with the intent of identifying optimal N requirements. Those that have, have found that in DS systems where the N is applied just before the permanent flood yields are comparable to WS systems where the N is applied in the recommended range of N applications for California (Pittelkow et al., 2014). In such DS systems, urea would most likely be the fertilizer of choice, and although urea is more expensive than aqua-ammonia, given the higher aqua-ammonia rate that would be required to apply preplant, the diffence in price is at least partially offset. Futhermore, by appling N fertilizer just before the permanent flood, N_2O emissions would be reduced compared to those in this study. Research has further shown that CH_4 could be further mitigated in the DS system using AWD or a mid-season drain (LaHue et al., 2016; Runkle et al., 2019).

It is important to highlight that this study did not quantify CO_2 emissions, which likely were increased in the DS system during the dry down events with soil organic matter mineralization (Moyano et al., 2013). In a review of studies comparing AWD to continuously flooded systems in the Asian context, albeit with a limited number of studies that quantified soil organic carbon (SOC) stocks and CO_2 emissions in the context of CH_4 and N_2O offsets, Livsey et al. (2019) found an 18.6% reduction in the soil-to-atmosphere carbon flux considering all three gases and a 5.2% reduction in SOC stocks in the AWD treatments. If DS is to be considered a viable option for rice production, further studies should consider a broader carbon footprint budget that incorporates all of these factors, on an annual basis, and in the context of legacy management particularly as it relates to SOC stocks.

Conclusion

The research here and elsewhere suggests that DS is a promising rice cropping and water management practice for CH_4 emissions and subsequently GWP reductions relative to WS. Dry seeding is also a viable management alternative to mitigate threats to yields associated with herbicide resistant weed species with yield potentials similar to WS. Maintaining comparable yields in DS rice likely requires shifts in N application management compared to WS due to early season N losses. The options we present for N application alternaties include an increased pre-plant N fertilizer application rate to compensate for predicted early seson N losses or surface applied N just before the permanent flood. While the latter may help to reduce early season N_2O emissions, which made a relatively minimal contribution to the GWP in this study, N_2O from rice systems still presents an environmental concern. Further research for the fine tuning of DS rice in terms of environmental impact should include attempts to better understand these suggested shifts in N application management, and the annual impact of DS management on GWP.

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