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Single midseason drainage events decrease global warming potential without sacrificing grain yield in flooded rice systems

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ABSTRACT

Rice (*Oryza sativa* L.) cultivation is an important part of global food security, yet it is also responsible for a significant portion of agricultural greenhouse gas (GHG) emissions, particularly methane (CH_4). Midseason drainage of flooded rice fields can decrease CH_4 emissions, but the magnitude of CH_4 reduction and its effect on grain yield are variable due to variation in the timing and soil-drying severity of drainage across studies. Therefore, in this two-year study, we aimed to quantify the effect of timing and severity of a single midseason drainage event on seasonal GHG emissions and grain yields, compared to a continuously flooded (CF) control. Treatments varied in terms of soil-drying severity (low, medium, and high, corresponding to approximately 5, 8, and 12 days of drying, respectively) and the timing of when drainage events occurred (between 34–49 and 45–59 days after seeding, or roughly between tillering and panicle initiation). Soil moisture parameters (perched water table, volumetric water content, gravimetric water content (GWC), and soil water potential), soil mineral nitrogen, CH_4 and nitrous oxide (N_2O) emissions, grain yield, and yield components were all quantified. Midseason drainage reduced seasonal CH_4 emissions by 38–66%, compared to the CF control. Seasonal CH_4 emissions decreased with increasing drain severity, and for every 1% reduction in soil GWC during the drainage period, seasonal CH_4 emissions were reduced by 2.5%. The timing of drainage had no significant impact on CH_4 emissions. Emissions of N_2O were low (average = 0.035 kg N_2O-N ha^{-1}) and accounted for only 0.5% of the seasonal global warming potential (GWP) across all drainage treatments. Within each year, drainage did not significantly affect grain yield compared to the CF control. Additionally, midseason drainage reduced both GWP and yield-scaled GWP by approximately the same amount as seasonal CH_4 emissions, as N_2O emissions were minimal and yields were similar across treatments. These results indicate that midseason drainage may be a viable GHG mitigation practice in flooded rice systems with limited risk for yield reduction, however, this practice should also be further tested under a broad range of soil types and different environments to determine its widespread adoptability.

1. Introduction

Rice (*Oryza sativa* L.) is a major staple crop in much of the world, as it accounts for 21% of the average human caloric intake (Awika, 2011) and is the third leading crop worldwide in terms of area harvested per year at approximately 162 million hectares (FAO, 2019). Rice cultivation, however, is also a significant source of methane (CH_4) emissions (Horwath, 2011; Linquist et al., 2012b; Yan et al., 2003) and is responsible for an estimated 22% of all agriculturally related CH_4 emissions as well as 11% of total anthropogenic CH_4 emissions (Smartt et al., 2016). Production of CH_4 in rice systems is primarily the result of methanogenic fermentation of soil organic matter under flooded anaerobic conditions (Mosier et al., 1998). Subsequent transfer of CH_4 to the atmosphere in such systems occurs primarily through the plant aerenchyma and at lower amounts via ebullition (Le Mer and Roger, 2001). A number of management practices have been studied and shown to mitigate CH_4 emissions from rice systems, including adjustment of crop residue management (Bhattacharyya et al., 2012; Xu et al., 2000), application of pre-composted organic matter (Wang and Shangguan, 1996; Wassmann et al., 2000), differential tillage (Ahmad et al., 2009; Harada et al., 2007), crop rotation (Feng et al., 2013), fertilizer use (Linquist et al., 2012a) and placement (Adviento-Borbe and Linquist,

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2016), and alternative water management via non-continuous flooding (Runkle et al., 2018; Yagi et al., 1997; Yang et al., 2017).

Non-continuous flooding practices vary and are known by several different names, including intermittent irrigation, alternate wetting and drying (AWD) (Siopongco et al., 2013) and alternate submergence-nonsubmergence (Belder et al., 2004), all of which commonly include multiple soil-drying events per growing season. Alternatively, midseason drainage is a form of non-continuous flooding which employs a single dry-down event during the mid to late vegetative stage of rice growth. With regards to on-farm implementation, midseason drainage may be considered a more practical alternative to practices which employ multiple drainage events, as it requires less overall maintenance and less risk of yield loss, as Carrijo et al. (2017) and others have noted that multiple drainage events throughout the growing season can result in an overall cumulative negative effect on yield. Additionally, midseason drainage may be an especially practical option in California rice systems, where it can be difficult to manage multiple drainage events due to water-holding time requirements following pesticide applications.

With regards to mitigation of CH_4 emissions, Jiang et al. (2019) reported in a global meta-analysis that, on average non-continuous flooding of rice fields reduces seasonal CH_4 emissions by 53%, compared to continuous flooding. Single drainage events tend to result in less mitigation of CH_4 emissions (33%) than that of multiple drainage events (47%) (Jiang et al., 2019), however, some individual studies have reported that one drainage event can result in reductions of CH_4 emissions comparable to that of practices which employ several drainage events (Itoh et al., 2011; Towprayoon et al., 2005). Increased soil-drying severity during drainage periods also tends to lead to greater reductions in CH_4 emissions (Jiang et al., 2019), however, Balaine et al. (2019) reported that once a certain level of soil-drying is achieved, further drying may not continue to reduce CH_4 emissions and can lead to increased risk of yield reductions. With regards to timing of drainage events, some have reported that drainage earlier in the season may more effectively reduce seasonal CH_4 emissions, compared to late drainage (Islam et al., 2018; Tariq et al., 2017), due to the presence of early-season peaks of CH_4 emissions, which can often be attributed to decomposition of previous crop residues (Chidthaisong and Watanabe, 1997). Additionally, drainage during the mid to late vegetative stage of rice growth has the potential to target peak seasonal CH_4 emissions (Balaine et al., 2019).

While midseason drainage is an effective management practice for CH_4 mitigation in flooded rice systems (Cai et al., 2003; Smith and Conen, 2004), it (as well as other non-continuous flooding strategies) can also lead to increased nitrous oxide (N_2O) emissions (Zou et al., 2007). Introduction of aerobic conditions to the soil during midseason drainage periods can cause nitrification of fertilizer ammonium (NH_4^+) and denitrification losses to the atmosphere (Buresh et al., 2008), both of which can lead to increased emissions of N_2O (Bateman and Baggs, 2005). In particular, drainage of fields when soil mineral nitrogen (N) levels are high can lead to increased N_2O emissions, therefore, in order to minimize N_2O emissions, it is important to coordinate drainage timing such that soil mineral N levels during drainage are relatively low (LaHue et al., 2016). In general, increases in N_2O emissions from these practices are often offset by larger decreases in CH_4 emissions, and as a result midseason drainage is an effective management practice for reducing seasonal global warming potential (GWP) (Akiyama et al., 2005), a measure of the atmospheric heating capacity of greenhouse gases (GHG) relative to that of carbon dioxide (CO_2). If water and N are not carefully co-managed in non-continuously flooded rice systems, N_2O emissions can, however, more than offset decreases in CH_4 emissions (Kritee et al., 2018; Lagomarsino et al., 2016).

With regards to the effect of non-continuous flooding practices on rice grain yield, results are quite variable. The higher the soil-drying severity of drainage events, the greater the risk for yield reductions, while drainage during reproductive and/or vegetative growth of rice generally has no significant impact on grain yield (Carrijo et al., 2017). In general, non-continuous flooding has been shown to decrease grain yield by only 3.6% on average, however, single drainage events have a less severe impact on yield than practices which employ two or more drainage events (Jiang et al., 2019). Upon examination of individual studies, some have reported that non-continuous flooding decreases grain yield (Linquist et al., 2015; Xu et al., 2015), has no effect (Lu et al., 2000; Pandey et al., 2014; Yao et al., 2012), or increases grain yield (Li, 2001; Liu et al., 2013), compared to continuous flooding. This variability is due to a number of factors but primarily because non-continuous flooding is implemented in a variety of ways on different soil types while utilizing different rice varieties, leading to variability in results. Additionally, drainage timing and soil-drying severity can vary considerably across studies. In many studies, specifics on water management are often not fully described, and information regarding timing, duration, and severity of drainage periods can be limited or missing.

In California, rice is primarily grown in the Sacramento Valley. A number of studies have evaluated the effects of non-continuous flooding on GHG emissions and rice grain yield in this region (Balaine et al., 2019; LaHue et al., 2016), but none to date have looked at how single midseason drainage events of varying soil-drying severity and timing affect these same variables. Additionally, these studies have examined AWD, which, from a practical standpoint, may be more difficult to employ in the field than single midseason drainage events. Furthermore, while the afore-mentioned meta-analyses (Carrijo et al., 2017; Jiang et al., 2019) allow for broad comparison of different water management practices and their effects on GHG emissions and yield, these comparisons are based on results from many studies which were limited in terms of the number of different water management practices evaluated side-by-side. Results stemming from the research in this study will be valuable in the continued development of alternative water management practices which can both mitigate GHG emissions and maintain grain yield, compared to a continuously flooded system.

In this two-year study, we aimed to further refine the development of sustainable water management practices for growing rice in the Sacramento Valley of California under non-continuously flooded conditions. Through the utilization of a single midseason drainage event, our broad objective was to reduce seasonal GHG emissions while maintaining grain yield, compared to a continuously flooded (CF) control. Our specific objective was to quantify the effect of both drain severity and timing on GHG emissions and yield. We hypothesized that increased soil-drying severity during the drainage period would significantly reduce seasonal CH_4 emissions compared to the CF control. With regards to drainage timing, we hypothesized that earlier midseason drainage would increase N_2O emissions while more effectively targeting peak seasonal CH_4 emissions, compared to late drainage.

2. Materials and methods

2.1. Study site description

This study was conducted at the Rice Experiment Station (39°27'47"N, 121°43'35"W) in Biggs, CA from April 2017 to October 2018, encompassing two growing seasons. Historical management of the research plots from 2012 to 2016 consisted of continuously flooded rice production, fields managed under AWD during growing seasons (treatment plots were randomized each year during this period), and flooding during the winter fallow periods (Balaine et al., 2019; LaHue et al., 2016). The soil at the site is a Vertisol, comprised of fine, smectitic, thermic, Xeric Epiaquerts and Duraquerts, with a soil texture of approximately 29% sand, 26% silt and 45% clay, a pH of 5.3, 1.06% organic C and 0.08% total N (Pittelkow et al., 2012). The climate at the site is Mediterranean with a mean annual precipitation of 444 mm and average daily temperatures of 17.8 °C (CIMIS Biggs, 2019). The total precipitation and average daily temperature during each growing season was 22.0 mm and 26.7 °C in 2017 and 11.2 mm and 24.4 °C in 2018 (CIMIS Biggs, 2019), respectively.

2.2. Treatments and experimental design

The entire experimental field size was 3.6 ha, and treatment plots were comprised of 0.3 ha basins, which were precision-leveled with no slope and separated by levees. Drain ditches between the plot edges and levees were constructed to help prevent seepage from adjacent plots. Treatment position within the experimental field was re-randomized each year. In both years, treatments were arranged in a randomized complete block design (RCBD) with three blocks and three replicates per treatment.

In each year of the study, three midseason drainage treatments were tested, which varied in timing and severity and were compared to a CF control. Treatment name prefixes were termed "E" (Early) or "L" (Late) according to when drainage periods occurred relative to each other. Drainage periods for all early treatments occurred between 34–49 days after seeding (DAS), while the drainage period for the one late treatment occurred between 45–59 DAS. Across all treatments, this timeframe spanned roughly between tillering and panicle initiation (PI). Treatment name suffixes corresponded to the intended soil-drying severity of each individual treatment (LS, MS, HS being Low, Medium and High severity, respectively). For example, an early low severity drain was termed "E-LS." In 2017, all midseason drainage treatments were early but varied in terms of severity (E-LS, E-MS, and E-HS). In 2018, treatments were identical to that of 2017 except for the inclusion of L-HS in lieu of E-MS.

In the CF control, the plots were flooded from immediately after sowing and remained flooded until roughly 3 weeks before harvest. For each drainage treatment a single drying period was imposed in which the irrigation was interrupted, and the floodwater was allowed to subside until there was no standing water. A drainage period was assumed to begin when the soil had no standing flood water but was still fully saturated. Plots were allowed to dry for a certain number of days and then reflooded depending on the treatment. The LS, MS and HS plots were allowed to dry for an average of 5, 8 and 12 days, respectively before being reflooded. The number of soil-drying days used in this study was determined based on a previous study at this location (Balaine et al., 2019) in which roughly 8 and 11 days of soil-drying resulted in soil volumetric water content (VWC) in the top 15 cm. of the soil to reach approximately 35% and 25%, respectively. The E-LS plots were reflooded 5 days after drainage, and this treatment was intended as a higher soil-drying severity alternative to "Safe-AWD" practices of previous studies (Bouman, 2007; Lampayan et al., 2015). In 2017, plots for each treatment were drained at different times but reflooded at the same time. In 2018, plots were both drained and reflooded at different times according to the individual treatments. In 2017, two plots from different treatments (E-LS and E-MS) did not dry down as intended due to seepage of water into the plots, despite the presence of drain ditches constructed to prevent seepage. Additionally, in 2018, it appeared that plots of the L-HS treatment were drying slowly based on VWC readings, thus these plots were allowed to dry an additional three days (14 days total) before being reflooded. However, we later determined that the VWC sensors in the high soil-drying severity plots were not reading accurately in either year. How the afore-mentioned plots were handled statistically is discussed in the data analysis section.

With the exception of the duration of the drainage periods, irrigation, nutrient, and pest management for drainage treatments was identical to that of the CF control. All drainage treatments were reflooded either shortly before or after PI, and in both years there was no precipitation during the drying periods. Detailed treatment and management information can be found in Table 1.

2.3. Field management

In both years of the experiment, 168 kg ha^{-1} of N fertilizer was applied as aqua-ammonia and injected at a soil depth of 7−10 cm prior to planting. All plots received 45 kg ha^{-1} of P_2O_5 (triple super phosphate) and 28 kg ha^{-1} of K_2O (muriate of potash) broadcast by plane approximately 4 weeks after planting. In order to assess soil mineral N status during drainage periods, 0 N subplots (3 by 6 m) were established in each main plot, and the location of these subplots was changed between years. After N fertilization, fields were direct seeded with the rice variety M-206 at the standard grower rate of 168 kg ha^{-1} by broadcasting seed directly onto the soil surface. Immediately after seeding, fields were flooded, and pests were controlled as necessary utilizing typical practices for CA rice growers. About 3 weeks before harvest all plots were drained and allowed to dry in preparation for harvest.

Prior to the 2017 growing season, rice straw from the 2016 harvest was chopped and incorporated into the soil, followed immediately by winter-flooding until mid to late February of 2017. After the 2017 growing season, rice straw was burned immediately after chopping, and fields were not intentionally flooded during the winter fallow period.

2.4. Soil moisture measurements

Soil moisture parameters are differentially affected by various physical soil factors such as texture, therefore four different soil moisture parameters were monitored during the drainage periods, including gravimetric water content (GWC), soil water potential (SWP), perched water table (PWT), and VWC. At 0–15 cm depth, VWC was monitored in both years using capacitance sensors (10HS, Decagon Devices Inc., Pullman, WA) connected to data loggers (Em50, Meter Group Inc., Pullman, USA). Units for VWC were measured as the ratio of water volume to soil volume expressed as a percentage. The sensors were installed vertically in the soil with the centers at a soil depth of 7.5 cm, and have a volume of influence of 1 L, which span from 0.5–14.5 cm soil depth. VWC was measured every 60 min. in 2017 and every 30 min. in 2018. SWP at 0–15 cm depth and 15−30 cm depth was monitored in both years using electrical resistance sensors (Watermark 200SS, Irrometer Co Inc., Riverside, CA). Units for SWP were measured in kilopascals (kPa). In 2017 SWP was measured once daily throughout the drainage periods. In 2018 electrical resistance sensors were connected to data loggers (900 M Monitor, Irrometer Co Inc., Riverside, CA), and SWP was measured every 60 min. The sensors were installed vertically in the soil with the centers at soil depths of 7.5 cm and 22.5 cm. One VWC sensor and two SWP sensors were installed in all drainage treatment plots and CF plots. PWT was measured at the end of the drainage period in all drainage treatment plots using perforated tubes. In each drainage treatment plot, a 60 cm long, 5 cm diameter polyvinyl chloride tube perforated with 1 cm diameter holes spaced approximately 2 cm apart was inserted 50 cm deep into the soil after drilling a hole of the exact same diameter. Soil GWC was measured immediately before each reflooding event for all drainage treatments. Soil GWC was determined by taking seven samples per plot to a depth of 30 cm using a 1.7 cm diameter soil core. Samples were sectioned and pooled into two soil depths (0–15 cm and 15–30 cm) and dried at 105 °C until constant weight was achieved. Soil GWC (%), was calculated as in Eq. (1):

$$GWC = 100 \text{ x } [(W-D)/D] \tag{1}$$

where: W = sample wet weight (g), D = sample dry weight (g).

2.5. Soil mineral nitrogen analyses

Soil (0−15 cm) mineral N (NH_4^+ and NO_3^-) was determined in both years from the 0 and 168 kg N ha^{-1} subplots in each main plot. In 2017, soil samples were taken weekly, starting approximately seven days after planting until immediately before drainage plots were reflooded. In 2018, soil samples were taken from each drainage treatment plot immediately before drainage and reflooding. Samples were homogenized and stored on ice for approximately 24 hours before measurement

Table 1
Dates for key management practices and crop stages for each year. DAS (days after seeding) was used to differentiate between timing of drainage events.

Management practice/growth stage	2017		2018	
	Date	DAS	Date	DAS
Fertilization	May 30	−4	May 20	−5
Sowing and initial flood	June 3	0	May 25	0
Propanil application	–	–	July 10	46
Panicle initiation (PI)	July 26	53	July 18	54
50% heading	Aug 16	74	Aug 8	75
Pre-harvest drain	Sep 17	106	Sep 16	114
Harvest	Oct 13	132	Oct 5	133
E-HS drained*	July 7	34	July 1	37
E-MS drained*	July 10	37	–	–
E-LS drained*	July 13	40	July 8	44
L-HS drained*	–	–	July 9	45
E-MS reflooded	July 18	45	–	–
E-HS reflooded	July 18	45	July 12	48
E-LS reflooded	July 18	45	July 13	49
L-HS reflooded	–	–	July 23	59

*Start dates for each drainage treatment refer to when soil from plots had no standing water but was fully saturated. E-HS = early, high severity; E-MS = early, medium severity; E-LS = early, low severity; L-HS = late, high severity.

of total soil mineral N. Three sub-samples (20 g) of each soil sample were added to 100 mL of 2 M KCl and mixed for one hour on a mechanical shaker. The solution was filtered through Whatman No. 42 filter paper (GE Healthcare UK Limited, Buckinghamshire, UK) and stored at 4 °C prior to colorimetric analysis for NH_4^+ (Verdouw et al., 1978) and NO_3^- (Doane and Horwath, 2003) on a spectrophotometer.

2.6. Greenhouse gas measurements and flux calculations

GHG samples were taken weekly throughout the growing seasons, however, during periods in which emission fluxes were expected to change rapidly (e.g. flooding and drainage events), sampling was conducted in consecutive days or every other day. Gases were captured in cylindrical flux chambers, which consisted of a permanent base installed prior to each field season, a variable-height extension to accommodate rice plant growth, and a sealed chamber with a vent tube for pressure equalization (Adviento-Borbe et al., 2013; Pittelkow et al., 2012). Gas samples (25 mL) were taken through silicon septa at 21, 42, and 63 min. and injected into pre-evacuated 12.5-mL glass vials (Labco Ltd., Buckinghamsire, UK). Four representative ambient gas samples were also taken at 0 min. during each sampling event. Gas sampling was conducted between 9:00 am and 12:00 pm PST, as soil temperatures and gas fluxes during this time period are expected to be representative of their average daily values (Adviento-Borbe et al., 2013). In order to reduce the effects of intensive gas sampling on rice plants, two collars were installed per main plot, and sampling alternated between collars for each sampling event. Boardwalks were established prior to each field season to minimize soil compaction and to prevent artificially inflated flux values.

All gas samples were analyzed for CH_4 and N_2O peak area on a GC-2014 gas chromatograph equipped with a ^{63}Ni electron capture detector (ECD) set at 325 °C for N_2O concentrations and a flame ionization detector (FID) for CH_4 concentrations (Shimadzu Scientific, Inst, Columbia, MD, USA). N_2O was separated by a stainless-steel column packed with Hayesep D, 80/100 mesh at 75 °C. The detection limits of the GC instrument were 1.83×10^{-4} mg L^{-1} for both CH_4 and N_2O. Results of the GC analyses were accepted if voltage output produced a linear relationship with the gas concentrations of CH_4 and N_2O standards with $r^2 > 0.99$ (1, 3.05, and 9.95 ppm for N_2O; 1.8, 10.18, 19.7, 100, 503, and 1020 ppm for CH_4); the peak area for each gas sample was then converted to a concentration based on this linear relationship. Fluxes were estimated from the linear increase of gas concentration over time, and gas concentrations were converted to elemental mass per unit area (g ha^{-1} d^{-1}) using the Ideal Gas Law with the chamber volume measured at each sampling event, the chamber air temperature measured as each gas sample was taken, and an atmospheric pressure of 0.101 MPa. Fluxes of CH_4 and N_2O were computed as:

$$F = (\Delta C/\Delta t)*(V/A)*\alpha \quad (2)$$

where F is the gas flux rate (g N_2O-N or CH_4-C ha^{-1} d^{-1}), $\Delta C/\Delta t$ denotes the increase of gas concentration in the chamber (g L^{-1} d^{-1}), V is the chamber volume (L), A is area covered by the chamber (ha), and α is a conversion coefficient for elemental C (α = 0.749) or N (α = 0.636). Individual flux values were integrated across all time points via linear interpolation to calculate cumulative seasonal emissions (spring tillage to harvest). As in other similar studies (Adviento-Borbe et al., 2013; Linquist et al., 2015; Pittelkow et al., 2012), gas fluxes with a linear correlation below a predetermined threshold ($r^2 = 0.9$) were treated as missing data, and those that were below the GC detection limits were set to zero flux for data analysis.

2.7. Yield and yield components

At physiological maturity, a small plot combine (2.18 m wide) was used to harvest four sample areas of approximately 13 m^2 within each drainage treatment plot. The sample areas were at least 5 m from the border of the plot. Grain moisture was measured for each sample, yields were corrected to 14% moisture, and the average of the four samples was considered the plot yield. Yield components were obtained by manually harvesting a 1 m^2 subplot and subsampling approximately 20% of the fresh biomass. The number of spikelets per panicle and percentage of unfilled grains per panicle were obtained from 15 panicles representative of the subsample. Grains were oven-dried at 65 °C until constant weight was achieved, weighed, and adjusted to 14% moisture for the estimation of yield. Straw was also oven-dried at 65 °C until constant weight and weighed, and harvest index was obtained as the mass ratio of grain to total aboveground biomass. Grain size was determined by weighing 1000 grains and correcting for 14% moisture. The total number of tillers was counted, and the number of panicles were counted from 50 representative tillers to estimate the percentage of unproductive tillers. Grain and straw were dried at 65 °C until constant weight for the determination of yield and harvest index.

2.8. Data analysis

Cumulative seasonal GWP was calculated for a 100-yr time horizon using radiative forcing potentials with climate-carbon feedbacks relative to CO_2 of 28 and 265 for CH_4 and N_2O, respectively (Pachauri et al., 2014). Yield-scaled GWP was calculated as the ratio of growing season GWP (kg CO_2-eq ha^{-1}) to grain yield (Mg ha^{-1}). We used R studio software (R Core Team, 2019) for analysis of variance on cumulative seasonal CH_4 emissions, N_2O emissions, and GWP with a protected Fisher's LSD means separation. The same was done for analysis of variance and means separation on rice grain yield, yield components, and yield-scaled GWP. In each instance, dependent variables were analyzed separately for each year due to the possibility of significant year by treatment interaction. Drainage treatment management was included as fixed effects, block by treatment interaction was included as random effects, and other interactions were included to the degree that it minimizes the corrected Akaike information criterion. In 2017, one plot was not included in yield analysis due to high weed infestation (not related to a treatment effect), and two additional plots from different treatments (E-LS and E-MS) were not included in soil moisture, yield, and daily/cumulative GHG emissions analysis because these plots did not dry down as intended according to the treatments. In order to quantify the effect of soil-drying severity on CH_4 emissions, simple

Table 2
Average grain yield (Mg ha^{-1}) for all treatments during the 2017 and 2018 growing seasons. Numbers in parentheses represent the standard errors of the mean. For each row, means followed by the same letter are not significantly different at $P < 0.05$.

		2017		
CF	E-LS	E-MS	E-HS	L-HS
9.06 a (0.33)	9.06 a (0.01)	8.85 a (0.50)	8.50 a (0)	–
		2018		
10.51 a (0.05)	10.38 a (0.16)	–	10.42 a (0.07)	10.82 a (0.25)

Table 3
Soil (0–15 cm) moisture parameters [gravimetric water content (GWC), soil water potential (SWP), perched water table (PWT), and volumetric water content (VWC)] measured just before reflood are shown. Sensors for VWC failed to function properly in HS plots and therefore data is not reported here (n/a). Data for the continuously flooded (CF) control represent seasonal averages. Numbers in parenthesis represent standard errors of the mean.

	2017				2018			
Treatment	GWC	SWP	PWT	VWC	GWC	SWP	PWT	VWC
	%	kPa	cm	%	%	kPa	cm	%
CF	48 (0)	0	+13	48 (1)	48 (1)	0	+13	49 (2)
E-LS	38 (2)	−15 (0)	−40 (1)	47 (1)	37 (2)	−6 (3)	−34 (3)	45 (0)
E-MS	29 (4)	−52 (7)	−37 (1)	36 (2)	–	–	–	–
E-HS	24 (4)	−96 (54)	−40 (5)	n/a	19 (1)	−94 (7)	−54 (4)	n/a
L-HS	–	–	–	–	22 (2)	−95 (16)	−58 (5)	n/a

linear regression analyses of soil drying parameters and seasonal CH_4 emissions were performed in Excel. The VWC data from the HS treatments was not used in this analysis because the sensors did not function at drier soil moisture levels.

3. Results

3.1. Grain yield and yield components

Rice grain yields of drainage treatments were not significantly different than that of the CF control in each year (Table 2). Additionally, yield components such as harvest index (HI), number of panicles per m^2, and grain weight were not significantly different than that of the CF control within each year (Supplemental Tables 1 and 2). Overall, yields in 2018 (average = 10.53 Mg ha^{-1}) were higher than in 2017 (average = 8.87 Mg ha^{-1}).

3.2. Soil moisture at the end of the drainage period

As expected, the drainage period decreased the values of all soil moisture parameters compared to that of the CF control (Table 3). The LS, MS, and HS treatments, on average, decreased soil GWC to 38%,

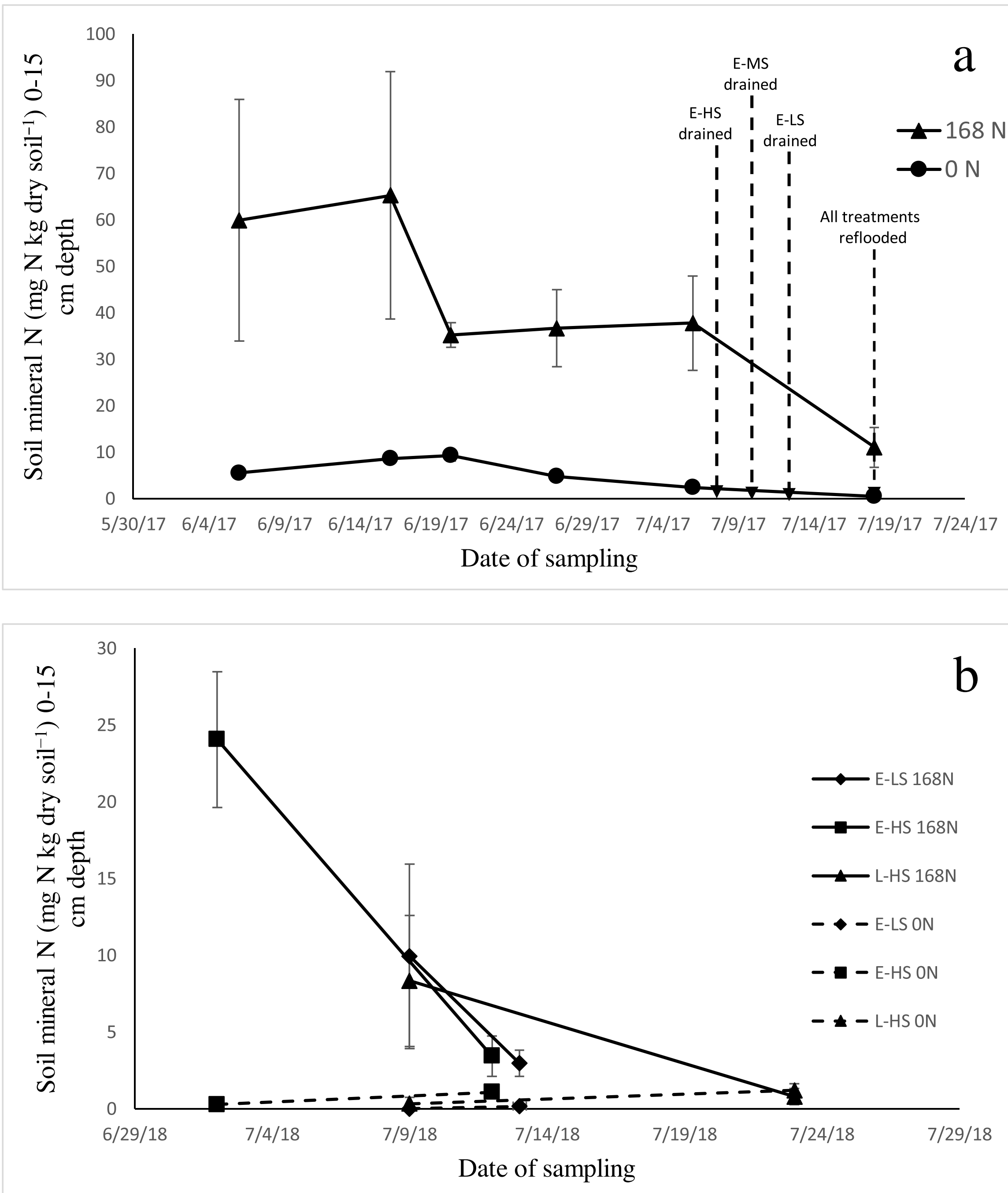

Fig. 1. Total soil mineral N (NH_4^+–N and NO_3^-–N) content at 0–15 cm depth during the 2017 (a) and 2018 growing season (b) in 0 N and 168 N subplots for all main plot water management treatments. Datapoints in (a) represent average values for all main plot water management treatments, while dashed vertical lines indicate when drainage periods began for each treatment and when plots were reflooded. In (b), sampling occurred immediately before drainage and reflood for each treatment. Fertilizer N was applied on 5/30 in 2017 (a) and on 5/20 in 2018 (b).

29%, and 22%, respectively, compared to the CF control, which averaged 48% throughout the growing season. The PWT reached an average of 37, 37, and 47 cm below the soil surface in LS, MS and HS treatments, respectively, over the two years, while these treatments also decreased SWP to an average of −11, −52, and −95 kPa, respectively, compared to 0 kPa in the control. Average VWC of the control was approximately 49% and decreased to an average of 46% and 36% in the LS and MS treatments (HS data not available), respectively over the two years of the study.

3.3. Soil mineral nitrogen

Soil mineral N (NH_4^+–N and NO_3^-–N) was measured in 0 N and 168 N subplots for all treatments to determine if N_2O emissions during drainage periods may be related to levels of soil mineral N at the time of drainage. In 2017, soil mineral N levels of the 0 N subplots were low just before drainage (2.44 mg N kg dry soil^{-1}) and just before reflood (0.49 mg N kg dry soil^{-1}) (Fig. 1a). Soil mineral N in the 168 N subplots declined from 38 mg N kg dry soil^{-1} just before drainage to 11 mg N kg dry soil^{-1} before reflood. By the time of reflood, soil mineral N in all main plot drainage treatments was relatively close to that of the 0 N subplots, compared to earlier in the season when it reached more than 60 mg N kg dry soil^{-1}.

In 2018, soil mineral N levels were measured for all treatments immediately before drainage and reflood of both 0 N and 168 N subplots in order to assess soil mineral N status during the drainage periods. In the 0 N subplots, soil mineral N levels were low just before drainage and before reflood, ranging between 0.26 and 1.18 mg N kg dry soil^{-1} (Fig. 1b). In the 168 N subplots of the E-HS treatment, soil mineral N levels were relatively high just before drainage (24 mg N kg dry soil^{-1}), compared to the other treatments. Soil mineral N during the drainage periods was low for E-LS and L-HS, ranging between 0.8–9.9 mg N kg dry soil^{-1} (Fig. 1b).

3.4. Methane emissions

3.4.1. Methane flux

In 2017, average daily CH_4 flux of the CF control peaked at approximately 4,729 g CH_4-C ha^{-1} day^{-1} (Fig. 2). Maximum daily CH_4 flux of the control occurred approximately 34 days after flooding, upon which the first drainage treatment began (E-HS). Average daily CH_4 flux of all drainage treatments declined immediately upon drainage of plots and declined to zero during the drying periods. Additionally, average daily CH_4 flux of E-HS remained particularly low for the remainder of

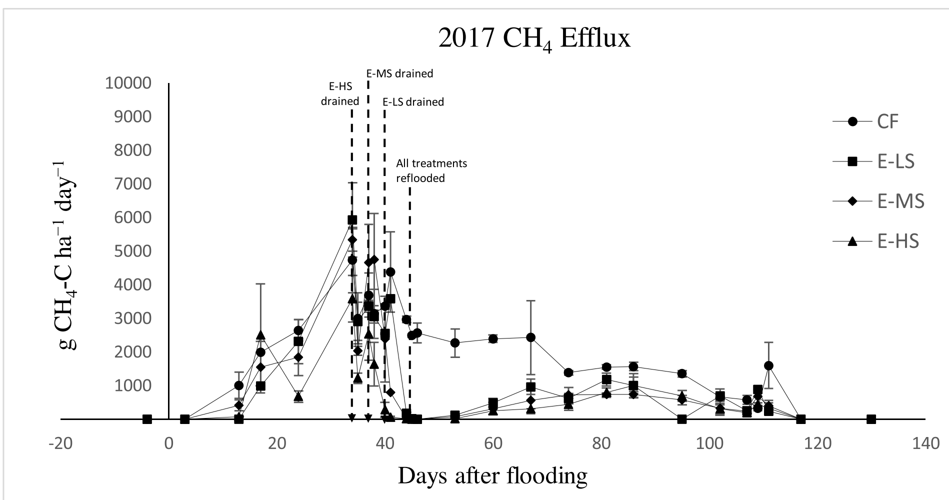

Fig. 2. Daily CH_4 emissions for each treatment in 2017. Dashed vertical lines indicate when drainage periods started and ended for each treatment. Error bars represent the standard errors of the mean. Seeding and initial flooding occurred on the same day.

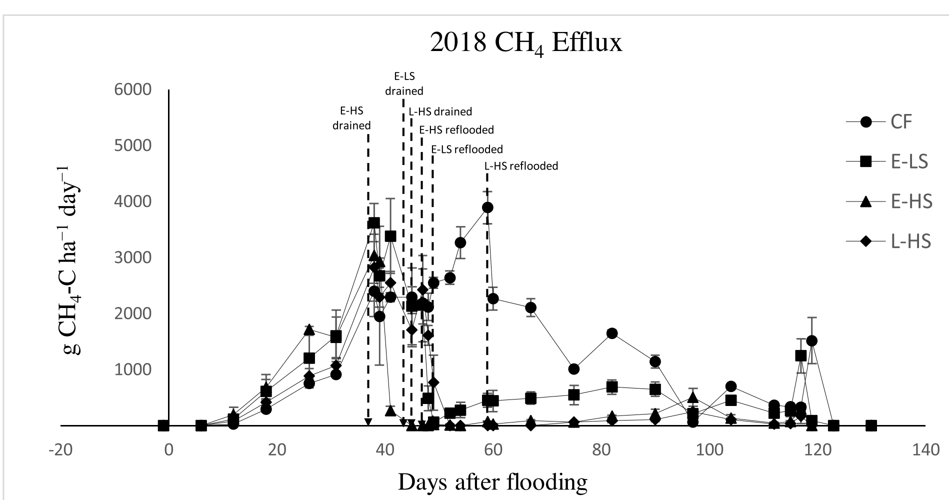

Fig. 3. Daily CH_4 emissions for each treatment in 2018. Dashed vertical lines indicate when drainage periods started and ended for each treatment. Error bars represent the standard errors of the mean. Seeding and initial flooding occurred on the same day.

the season, never surpassing 1,000 g CH_4-C ha^{-1} day^{-1} (Fig. 2). Between 36–41 days after reflood, average daily CH_4 flux of both E-LS and E-MS treatments reached approximately half of that of the CF control. In the control, a relatively large spike in CH_4 emissions was observed during the end-of-season drain. This spike represented a maximum daily CH_4 flux of approximately 1,592 g CH_4-C ha^{-1} day^{-1} in the control, while all drainage treatments resulted in comparatively low end-of-season CH_4 spikes (average = 668 g CH_4-C ha^{-1} day^{-1}).

In 2018, two peaks of average daily CH_4 flux in the CF control occurred at 38 and 59 days after flooding (Fig. 3). As was the case in 2017, average daily CH_4 flux decreased immediately upon drainage of plots. E-HS and L-HS decreased daily CH_4 emissions to zero during the drying periods, while E-LS did not. Average daily CH_4 flux of E-HS and L-HS remained low throughout the season after reflood (Fig. 3). The maximum average daily CH_4 flux of E-HS and L-HS post-treatment was only 500 and 244 g CH_4-C ha^{-1} day^{-1}, respectively, while average daily CH_4 flux of E-LS remained close to that of the CF control during the post-treatment period of the growing season. For both E-LS and the CF control, a relatively large spike in CH_4 emissions was observed during the end-of-season drain. Daily CH_4 flux of approximately 1,245 g CH_4-C ha^{-1} day^{-1} and 1,514 g CH_4-C ha^{-1} day^{-1} constituted these spikes in E-LS and CF, respectively, while comparatively smaller end-of-season spikes were seen in the two high-severity treatments (average = 229 g CH_4-C ha^{-1} day^{-1}).

3.4.2. Cumulative methane emissions

Midseason drainage treatments across both years of the study reduced cumulative seasonal CH_4 emissions by between 38–66% compared to the CF control. Specifically, high severity drainage treatments such as E-HS and L-HS reduced seasonal CH_4 emissions by an average of 64%, compared to the control, while lower severity treatments such as E-LS and E-MS reduced seasonal CH_4 emissions by an average of 38% and 46%, respectively. There were no significant differences in seasonal CH_4 emissions between E-HS and L-HS during 2018 (Table 4). Additionally, average cumulative seasonal CH_4 emissions decreased by 34% from 2017 to 2018 across all treatments.

3.4.3. Soil moisture vs. cumulative methane emissions

Upon examination of the relationship between soil moisture at the end of the drainage period and cumulative seasonal CH_4 emissions, GWC was most strongly related to cumulative CH_4 emissions with an average linear R^2 of 0.67 across both seasons. Within each year of the study,

Table 4
Cumulative seasonal CH_4 and N_2O emissions, GWP and yield-scaled GWP for each year and treatment. Numbers in parentheses represent the standard errors of the mean. For each column within each year, means followed by the same letter are not significantly different at $P < 0.05$.

Treatment	CH_4	N_2O	GWP	Yield-scaled GWP
	kg CH_4-C ha^{-1}	kg N_2O-N ha^{-1}	kg CO_2 eq ha^{-1}	kg CO_2 eq Mg^{-1}
	2017			
CF	212 (15) a	0 (0) a	7913 (576) a	873 (59) a
E-LS	132 (4) b	0 (0) a	4911 (154) b	542 (17) b
E-MS	114 (13) bc	0.05 (0.02) a	4253 (478) bc	482 (26) b
E-HS	80 (19) c	0.16 (0.07) a	2971 (695) c	426 (66) b
	2018			
CF	153 (8) a	0 (0) a	5700 (289) a	542 (27) a
E-LS	94 (16) b	0 (0) a	3512 (585) b	340 (60) b
E-HS	52 (7) c	0 (0) a	1957 (275) c	188 (27) c
L-HS	55 (6) c	0 (0) a	2059 (230) c	191 (24) c

Table 5
Results of simple linear regressions between three soil (0–15 cm) moisture parameters [gravimetric water content (GWC), soil water potential (SWP), and perched water table (PWT)] immediately before reflood and cumulative seasonal CH_4 emissions (kg CH_4-C ha^{-1}), including linear R^2 values, standard errors, and P-values. P-values <0.05 indicate significant differences at 95% confidence interval.

	2017			2018		
	GWC	SWP	PWT	GWC	SWP	PWT
R^2	0.56	0.51	0.38	0.78	0.55	0.85
Standard Error	53.08	56.34	62.93	22.10	31.51	17.93
P-value	<0.01	<0.01	0.03	<0.001	<0.01	<0.0001

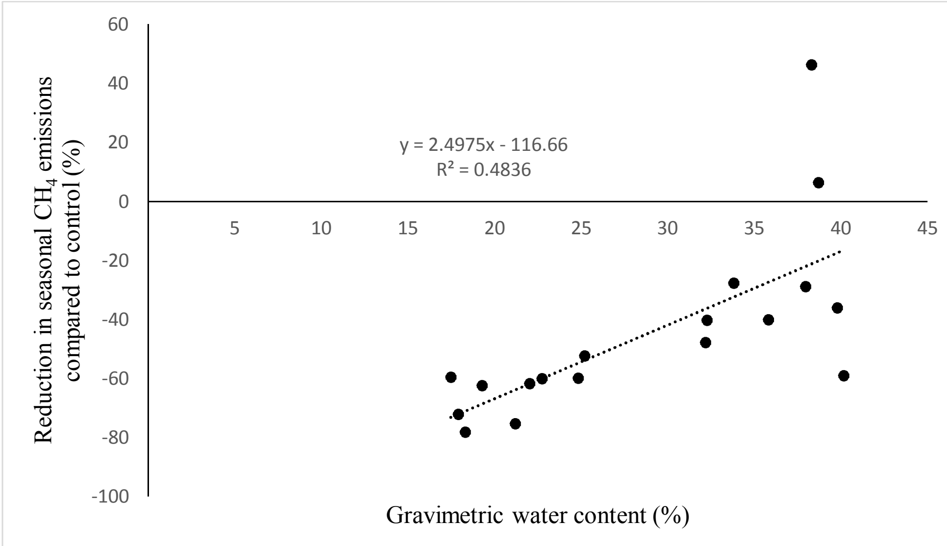

Fig. 4. Simple linear regression between soil gravimetric water content (%) just before reflood and reduction in seasonal CH_4 emissions compared to the CF control (%) from both 2017 and 2018. Seasonal CH_4 emissions value used for the CF control in each year was the mean value of all CF plots in each given year.

three soil moisture parameters resulted in linear R^2 values between 0.38 and 0.85 (Table 5).

In order to account for annual variation in cumulative CH_4 emissions, a simple linear regression was performed for soil GWC just before reflood vs. percent reduction in seasonal CH_4 emissions compared to the CF control. Soil GWC explained approximately 48% of the variation in percent reduction of CH_4 emissions across all drainage treatment plots (Fig. 4). Furthermore, this relationship indicates that for every 1% reduction in GWC, seasonal CH_4 emissions are reduced by approximately 2.5%.

3.5. Nitrous oxide emissions

Cumulative seasonal N_2O emissions were low and averaged 0.035 kg N_2O-N ha^{-1} across all drainage treatments for each year and accounted for only 0.5% of the seasonal GWP for all drainage treatments. In 2018 in particular, N_2O emissions were all below the detection limit threshold of the GC, and therefore fluxes were considered to be zero for all treatments. In 2017, cumulative seasonal N_2O emissions were positive but low in E-MS and E-HS treatments (Table 4), and all such N_2O fluxes occurred during the drainage periods (data not shown).

3.6. GWP and yield-scaled GWP

Cumulative seasonal GWP was reduced for all drainage treatments compared to the CF control (Table 4), and reductions in GWP were almost identical to that of seasonal CH_4 emissions because N_2O emissions were low across all treatments. Similarly, reductions in yield-scaled GWP compared to the CF control were nearly identical to that of seasonal CH_4 emissions and GWP, as yields did not vary considerably within each year of the study.

4. Discussion

4.1. Grain yield

In 2017, average rice yields (8.87 Mg ha^{-1}) were lower than in 2018 (10.53 Mg ha^{-1}). This reflected similar patterns in average statewide rice yields in California between these two years and was attributed in part to abnormally high mid- and late-season temperatures during the 2017 growing season (Childs et al., 2018). Additionally, in neither year of the study did drainage significantly impact rice yields (Table 2). Carrijo et al. (2017) reported that yields under drainage conditions generally decrease when SWP ≤ -20 kPa. In this study, E-MS, E-HS, and L-HS all reached SWP values below this threshold (between -52 and -96 kPa) before reflooding (Table 3) and resulted in no significant yield reductions. This may be due to the ability of rice roots to grow into the shallow water table at this particular location, which can aid in water uptake during dry periods and was reported by Carrijo et al. (2018). It is important to note, however, that this level of soil-drying may not be suitable for growth of rice in all environments. At this study site in particular, the water table is relatively shallow and is likely a contributing factor in rice's ability to tolerate such drying conditions.

4.2. Methane emissions

4.2.1. Cumulative seasonal methane emissions

Cumulative seasonal CH_4 emissions varied between the two years of this study. Average seasonal CH_4 emissions from the CF control in 2017 were higher than that of the control in 2018 by approximately 59 kg CH_4-C ha^{-1}. Seasonal CH_4 emissions from the control in 2017 (average = 212 kg CH_4-C ha^{-1}) were near the upper limit of the 95% confidence interval for flooded California rice systems (114–213 kg CH_4-C ha^{-1}), based on a meta-analysis by Linquist et al. (2018), while seasonal CH_4 emissions from the control in 2018 (average = 153 kg CH_4-C ha^{-1}) were well within this range. Experiments conducted at the same location in previous years have also reported high seasonal CH_4 emissions in addition to variation in emissions from year to year (Balaine et al., 2019; LaHue et al., 2016). While the cause of year-to-year variation is often not clear, variation between 2017 and 2018 of this study, however, can be explained by the fact that all plots were burned following the 2017 season, which can decrease incorporated soil organic carbon from crop residues (Chan et al., 2002) and lower cumulative CH_4 emissions in the following season. High seasonal CH_4 emissions in the 2017 growing season may be due to high yields (and subsequently high straw biomass) reported at this location in the previous year (Balaine et al., 2019), as residues left in the field from the previous growing season can provide additional carbon substrate for methanogenic fermentation of soil organic matter (Yan et al., 2005).

4.2.2. Soil-drying severity and methane emissions

We hypothesized that increasing the severity of the drainage period would lead to increased reductions in seasonal CH_4 emissions. In agreement with this hypothesis, CH_4 emissions decreased by approximately 2.5% for every 1% decline in GWC during the drying period. The two high soil-drying severity treatments in this study (E-HS and L-HS) reduced seasonal CH_4 emissions, on average, by 64%, compared to the control. These reductions are higher than those reported in a U.S.-based meta-analysis by Linquist et al. (2018) for single dry-down events (average = 39%; 95% confidence interval 30–47%). They are also higher than those reported in a global meta-analysis by Jiang et al. (2019) for single dry-down events (average = 33%; 95% confidence interval 12–49%). The larger reductions in seasonal CH_4 emissions from E-HS and L-HS are likely due to the higher soil-drying severity of these treatments, compared to most studies included in the aforementioned meta-analyses. In support of this, CH_4 emissions reductions from E-HS and L-HS were within the 95% confidence interval for high severity drainage treatments studied globally (95% confidence interval 60–84%) (Jiang et al., 2019). The percent reductions we report from E-HS and L-HS are also similar to those reported by Itoh et al. (2011), in which longer midseason drainage periods were employed.

The average percent reduction in cumulative CH_4 emissions reported here using a single midseason drain of high soil-drying severity (64%) is the same as that reported by Balaine et al. (2019) in which two drainage events were tested at this same location in 2015 and 2016 with similar soil-drying severity. At this location, these high-severity AWD treatments tested in Balaine et al. (2019) reduced CH_4 emissions by approximately 64%, which suggests that more than one drainage event of high soil-drying severity may not be necessary for significant mitigation of seasonal CH_4 emissions. This may be due to strong oxidation of the soil during a single drainage event of high soil-drying severity, which can significantly decrease methanogenic activity both during the drainage period and after reflood (Ma and Lu, 2011). A single midseason drainage event is also more practical for growers, as farmers do not have to concern themselves with the excess water management that is associated with AWD practices. There is also less risk of yield losses with midseason drainage, compared to AWD, as multiple drainage events throughout the growing season have been shown to significantly decrease grain yield (Carrijo et al., 2017).

While the low and medium severity drainage treatments (E-LS and E-MS) significantly reduced cumulative seasonal CH_4 emissions relative to the control (Table 4), they were less effective than the two HS treatments. This is likely due to lower levels of oxidation of the soil during the drainage periods for E-LS and E-MS, allowing for higher daily CH_4 flux after plots had been reflooded (Figs. 3 and 4). Soil drying severity has been defined as "mild" if SWP is ≥ -20 kPa or if field water level does not drop below 15 cm from the soil surface (Carrijo et al., 2017; Jiang et al., 2019; Lampayan et al., 2015). The E-LS treatment of this study resembled mild soil-drying severity with regards to SWP, as it reduced SWP to -15 kPa and -6 kPa in 2017 and 2018, respectively; however, it decreased PWT to below the threshold value for mild soil-drying severity in each year. Importantly, the E-LS treatment is a slightly more severe alternative to what is referred to as "Safe-AWD" (Bouman, 2007; Lampayan et al., 2015), which has been shown to reduce CH_4 emissions with little risk of yield loss. In studies in which Safe-AWD has been tested, multiple soil-drying events have been found to reduce cumulative CH_4 emissions by 41% (Balaine et al., 2019), or, based on a global meta-analysis, by 56% (Jiang et al., 2019). In this study, with a single midseason drainage event, the E-LS treatment reduced cumulative seasonal CH_4 emissions by approximately 38%, compared to the CF control.

4.2.3. Drainage timing and methane emissions

A number of studies have reported that drainage earlier in the season may more effectively target seasonal CH_4 emissions than during midseason (Tariq et al., 2017; Islam et al., 2018). In flooded rice systems, a peak in CH_4 emissions often occurs relatively early in the season, which is primarily attributed to decomposition of the previous crop residues (Chidthaisong and Watanabe, 1997). Based on this, we hypothesized that E-HS, in which the drainage period began 37 DAS, would more effectively target an early season peak in CH_4 emissions and reduce CH_4 emissions more than L-HS, in which the drain was initiated 8 days later. While peak early-season CH_4 emissions were seen in both years (Figs. 3 and 4), cumulative CH_4 emissions were similar between E-HS and L-HS (Table 4). Reasons for the lack of difference between these treatments may be due to the relatively small difference in time (8 days) between the drains. Secondly, pre-drain CH_4 emissions for E-HS were higher than that of L-HS; and, while this was not a significant difference, it resulted in offsetting some of the intended benefit of draining early for the E-HS treatment. Efforts to target seasonal peaks in CH_4 emissions can be challenging because these peaks do not always occur early in the season as evidenced by Ahn et al. (2014) and Linquist et al. (2015). There are a number of factors which contribute to the timing of peak CH_4 emissions during the growing season such as temperature, management of the previous season's residue, soil type, and rice cultivar. In water-seeded systems such as the one in this study, in which the majority of fertilizer is applied before planting, draining particularly early in the season may risk significantly increasing N_2O emissions due to the high amount of N fertilizer present in the soil at the time of drainage (Kritee et al., 2018; LaHue et al., 2016) and the subsequent N losses to the atmosphere due to nitrification and denitrification (Buresh et al., 2008).

4.2.4. Best indicators of methane emissions

Many methods can be used to assess soil moisture status during drainage events and can help in determining when reflooding is necessary to prevent yield loss as well as providing an indicator of potential CH_4 mitigation. Simple linear regressions between soil-drying severity and seasonal CH_4 emissions showed that SWP, PWT, and GWC all accounted for a large portion of variation in seasonal CH_4 emissions across drainage treatments (Table 5). PWT as an indicator of seasonal CH_4 emissions displayed the largest variation between the two years of the study. The reason for this may be due to the fact that PWT decreased to at least 34 cm below the soil surface for all drainage treatments, and as PWT lowers it can become an increasingly poor indicator of surface soil moisture (Lampayan et al., 2015). Efflux of CH_4, however, is largely governed by the conditions of the soil surface layer, which has the greatest potential for both CH_4 production (due to high levels of organic matter) and oxidation (caused by aerobic conditions) (Conrad and Rothfuss, 1991; Xiao et al., 2017). In contrast to PWT, other soil moisture parameters evaluated in this study such as SWP and GWC were measured at the soil surface layers and were therefore more consistent in explaining variation in CH_4 emissions.

In general, the aforementioned soil moisture parameters vary across soil type and particularly across different soil textures. It was expected that VWC, in particular, would be best at explaining variation in cumulative seasonal CH_4 emissions as it is most directly related to soil oxygen availability or lack thereof (Feng et al., 2002). However, this was found not to be the case, and instead soil GWC, on average, was found to be best at explaining variation in seasonal CH_4 emissions across the two years of the study (Table 5). Importantly, in this study, we were not able to rigorously assess soil VWC because VWC readings in the field failed to decline with increasing soil dryness at high severity. A reason for this may be due to the development of cracks in the soil at this site (a Vertisol) during the drainage periods, which may have led to differential texture and/or flow paths of soil water and therefore inconsistent VWC readings in the field. As soil GWC was measured directly and the experimental plots had similar soil texture, it was found to be the most reliable assessment of soil moisture status and indicator of potential CH_4 mitigation.

4.3. Nitrous oxide emissions

Generally, N_2O emissions are low in flooded rice systems as most of the N_2O that is produced undergoes further reduction and is emitted as atmospheric N_2 (Firestone and Davidson et al., 1989). Midseason drainage, however, imposes soil conditions that can lead to N_2O

emissions, as soil-drying and reflooding causes nitrification and denitrification, respectively (Buresh et al., 2008), both of which are processes that can lead to production of N_2O (Dobbie et al., 1999). While a number of studies have reported increases in N_2O emissions as a result of midseason drainage (Zou et al., 2007), in the majority of instances reductions in CH_4 emissions more than offset the increase in N_2O emissions, which leads to decreases in seasonal GWP (Jiang et al., 2019).

In both years of our study, N_2O emissions were low and close to zero. In 2017, drainage resulted in small increases in N_2O emissions, which were not significantly different across all treatments (Table 4). These N_2O emissions occurred during the drainage periods only (data not shown). In 2017, E-HS resulted in the highest N_2O emissions of all drainage treatments, which may have been due to the fact that drainage for this treatment occurred earlier in the season than that of other treatments, when soil mineral N was relatively high (Fig. 1a). In 2018, it was expected that E-HS may lead to increased N_2O emissions compared to L-HS due to introduction of aerobic conditions at a time when soil mineral N is higher. Emissions of N_2O, however, during this year were especially low, and there were no significant differences in N_2O emissions between all treatments in 2018. The reason for this may also be due to relatively low soil mineral N levels at the time of soil-drying (Fig. 1b) for E-LS and L-HS treatments or N loss primarily in the form of atmospheric N_2 rather than N_2O during the drainage period of E-HS. Similarly, low N_2O emissions across these treatments from both years may also be explained by the fact that drainage of plots occurred during the late vegetative stage of rice growth around PI, which ensures that most fertilizer N has been taken up by the plants (Peng and Cassman, 1998), leaving little available soil mineral N to undergo nitrification and denitrification.

4.4. GWP and yield-scaled GWP

Cumulative seasonal GWP of the CF control was 7,913 and 5,700 kg CO_2-eq ha^{-1} in 2017 and 2018, respectively. These values are high relative to global flooded rice systems (average = 3,757 kg CO_2-eq ha^{-1}) (Linquist et al., 2012b) but less than that of other California rice systems (Linquist et al., 2018). One possible reason that emissions from this study are higher than that of global rice systems may be due to the high yields associated with this study, with high amounts of incorporated straw and organic matter leading to higher emissions in following seasons (Xu and Hosen, 2010). There is, however, tremendous variability associated with the global average for seasonal GWP of flooded rice systems (range = 75−22,237 kg CO_2-eq ha^{-1}) (Linquist et al., 2012b), therefore it is difficult to evaluate these differences with a high degree of certainty.

Given concerns with climate change and food security for an increasing global population, yield-scaled GWP may be considered a more appropriate metric for evaluation of mitigation treatments for sustainable intensification (van Groenigen et al., 2010), as it quantifies GWP in relation to grain yield. As CH_4 emissions were high and yields were lower than expected in 2017, yield-scaled GWP of the CF control (average = 873 kg CO_2-eq Mg^{-1}) was greater than the global average yield-scaled GWP for flooded rice systems (average = 657 kg CO_2-eq Mg^{-1}) reported by Linquist et al. (2012b). Due to decreased CH_4 emissions and relatively high yields in 2018 compared to the previous year, yield-scaled GWP of the CF control in 2018 (average = 542 kg CO_2-eq Mg^{-1}) was less than the reported global average.

All drainage treatments in this study reduced GWP and yield-scaled GWP by approximately the same relative amount as that of seasonal CH_4 emissions because N_2O emissions and yields were similar across treatments in each year. The two high severity drainage treatments, E-HS and L-HS, reduced yield-scaled GWP by an average of 57% and 65%, respectively, which are similar to reductions in yield-scaled GWP from AWD reported in several tropical environments (Oo et al., 2018; Pandey et al., 2014; Tariq et al., 2017). These reductions are also greater than those reported in a global meta-analysis by Jiang et al. (2019) with respect to single drainage events (average = 15%), but this is likely due to the fact that soil-drying severity of these treatments was greater than that of most other single dry-down treatments studied globally, as mentioned earlier. Additionally, percent reductions in yield-scaled GWP due to E-HS and L-HS were greater than that of most other high severity drainage treatments which have been studied globally (average = 45%) (Jiang et al., 2019).

With regards to low and medium severity drainage treatments in this study, E-LS and E-MS reduced seasonal GWP on average by 38% and 47%, compared to the CF control. While the E-LS treatment was similar to what is referred to as Safe-AWD or mild soil-drying severity (Carrijo et al., 2017; Jiang et al., 2019; Lampayan et al., 2015), the E-MS treatment actually closely resembled many high soil-drying severity treatments of various other studies in terms of soil moisture. The E-MS treatment accordingly resulted in reductions of seasonal GWP and yield-scaled GWP that are within the range of many high severity drainage treatments which have been studied. Globally, drainage periods of mild soil-drying severity reduce GWP by 26%, on average, however, this figure is highly variable (95% confidence interval: -64.5% to +53.3%) (Jiang et al., 2019). Thus, while drainage treatments of mild soil-drying severity pose less risk to yield reductions, GHG mitigation resulting from such treatments can be limited or highly variable. Importantly, while E-LS may be a safe option for maintaining grain yield and mitigating GHG emissions to a certain extent, soil-drying severity of E-MS may be too high for some regions without a relatively shallow water table. Additionally, reductions in yield-scaled GWP for both E-LS and E-MS were nearly identical to that of reductions in GWP, as yields were similar across treatments within each year.

5. Conclusions

In this experiment it was found that implementation of a single midseason drainage event has the potential to reduce seasonal cumulative CH_4 emissions from flooded rice cultivation while maintaining peak levels of production in California rice systems. Soil-drying severity was strongly related to percent reductions in CH_4 emissions, but the timing of the drainage periods in this study did not significantly impact emissions. Special attention, however, should be given to drainage timing and soil-drying severity, as drainage much earlier than was done in this study may increase N_2O emissions, and high soil-drying severity may be a potential risk for yield reductions. If water and N inputs are carefully co-managed and consideration is given to various site-specific characteristics, midseason drainage may be considered a safe and suitable GHG mitigation practice. Given the variability of CH_4 emissions due to different soil types and residue management, similar studies as this one should be conducted across a wide range of soil types, farmer management practices, and locations of differing water table depths to assess adoption of midseason drainage on a broader scale. While obstacles exist to the widespread adoption of non-continuous flooding practices as a whole, particularly in California, this research highlights the potential of such a practice to bring about significant environmental and agronomic benefits.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fcr.2021.108312.

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