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# Greenhouse Gas Emissions and Management Practices that Affect Emissions in US Rice Systems

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## Abstract

Previous reviews have quantified factors affecting greenhouse gas (GHG) emissions from Asian rice (*Oryza sativa* L.) systems, but not from rice systems typical for the United States, which often vary considerably particularly in practices (i.e., water and carbon management) that affect emissions. Using meta-analytic and regression approaches, existing data from the United States were examined to quantify GHG emissions and major practices affecting emissions. Due to different production practices, major rice production regions were defined as the mid-South (Arkansas, Texas, Louisiana, Mississippi, and Missouri) and California, with emissions being evaluated separately. Average growing season $CH_4$ emissions for the mid-South and California were 194 (95% confidence interval [CI] = 129–260) and 218 kg $CH_4$ ha$^{-1}$ season$^{-1}$ (95% CI = 153–284), respectively. Growing season $N_2O$ emissions were similar between regions (0.14 kg $N_2O$ ha$^{-1}$ season$^{-1}$). Ratoon cropping (allowing an additional harvestable crop to grow from stubble after the initial harvest), common along the Gulf Coast of the mid-South, had average $CH_4$ emissions of 540 kg $CH_4$ ha$^{-1}$ season$^{-1}$ (95% CI = 465–614). Water and residue management practices such as alternate wetting and drying, and stand establishment method (water vs. dry seeding), and the amount of residue from the previous crop had the largest effect on growing season $CH_4$ emissions. However, soil texture, sulfate additions, and cultivar selection also affected growing season $CH_4$ emissions. This analysis can be used for the development of tools to estimate and mitigate GHG emissions from US rice systems and other similarly mechanized systems in temperate regions.

## Core Ideas

- Emissions of $CH_4$ and $N_2O$ were quantified for US rice systems using a meta-analysis.
- Emissions were determined for both the growing and fallow seasons.
- We assessed the major management practices affecting emissions.
- Analysis can be used to develop a tool for quantifying emissions from rice fields.



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For agricultural systems, flooded rice (*Oryza sativa* L.) systems are relatively large emitters of greenhouse gases (GHG)—particularly due to high $CH_4$ emissions (Linquist et al., 2012b). There have been considerable efforts made toward broadly quantifying GHG emissions from rice systems, as well as quantifying the effects of major variables responsible for emissions. These efforts have focused on Asia (Akiyama et al., 2005; Yan et al., 2005, 2009), which produces ∼90% of the world's rice, and have been used to provide guidelines to quantify national GHG inventories (Eggleston et al., 2006). However, no extensive reviews have been conducted for US rice systems. The United States, with some of the highest grain yields in the world, is typically ranked around the 10th in total rice production and is in the top five countries for rice exports (FAO, 2017). Data from the USDA National Agriculture Statistics Service over the past decade indicate that roughly 80% of the rice grown in the United States is grown in the mid-South (Arkansas, Louisiana, Texas, Mississippi, and Missouri), whereas ∼20% is produced in California.

Given the complexity of GHG emissions, process-based models would be ideal for quantifying GHG emissions, as well as evaluating potential mitigation options. Models in various stages of development including DeNitrification-DeComposition (DNDC, http://www.dndc.sr.unh.edu/), Landscape DNDC (Haas et al., 2013), and DAYCENT (a daily version of the CENTURY biogeochemical model; Parton et al., 1993) are able to predict seasonal emissions acceptably. However, these models do not estimate daily fluxes well, suggesting that the underlying assumptions and processes controlling emissions are not fully understood or incorporated into the current models. This is a particular concern when quantifying the effects of mitigation options, as the various processes affecting emissions need to be fully understood in order for these models to provide reliable emissions estimates. Some examples of these modeling efforts for rice production include Simmonds et al. (2015b) for DNDC, Kraus et al. (2015) for Landscape DNDC, and Cheng et al.

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**Abbreviations**: AWD, alternate wetting and drying; CI, confidence interval; DNDC, DeNitrification-DeComposition; GHG, greenhouse gas; IPCC, Intergovernmental Panel on Climate Change.

(2013) for DAYCENT. To improve these models, Kraus et al. (2015) suggested that they be coupled to more complex hydrological models to integrate the complex interactions between the soil, atmosphere, and hydrosphere.

Another approach for quantifying emissions is that used by the Intergovernmental Panel on Climate Change (IPCC), which developed a protocol to estimate national inventories of GHG emissions (Eggleston et al., 2006). This methodology includes several tiers depending on data availability from any given country. At its simplest, the Tier-1 approach uses a baseline emission factor for rice and adjusts it using scaling factors for different management practices or situations. For rice $CH_4$ emissions, the IPCC (Eggleston et al., 2006) uses a baseline emission factor of 1.3 kg $CH_4$ $ha^{-1}$ $d^{-1}$ (ranging from 0.8–2.2 kg $CH_4$ $ha^{-1}$ $d^{-1}$). This emission factor was determined according to an analysis by Yan et al. (2005), who also provided the basis for the scaling factors used in the current IPCC methodology (Eggleston et al., 2006). The database used for their analysis used only sites from Asia, and the scaling factors were most appropriate to Asian rice production systems. For $N_2O$, the IPCC (Eggleston et al., 2006) default is 0.003 kg $N_2O$-N $kg^{-1}$ N applied, is based on work by Akiyama et al. (2005), and is also based on data primarily from Asia. This emission factor is lower than the 0.01 kg $N_2O$-N kg $N_2O$-N $kg^{-1}$ N applied used for upland crops (Eggleston et al., 2006).

The IPCC (Eggleston et al., 2006) guidelines suggest using other approaches to improve the accuracy of inventory estimates if local data are available. Recently, the USEPA used a combination of IPCC Tier-1 and Tier-3 approaches to estimate GHG inventories from US rice systems (USEPA, 2017). The Tier-3 approach used the DAYCENT model to estimate $CH_4$ emissions based on the work of Cheng et al. (2013) in rice fields in China. Although the USEPA approach provides a good basis to quantify GHG inventories at a national level, it is limited in that the DAYCENT model is not able to accurately quantify changes in emissions relative to changes in management practices, as discussed above.

Previous reviews (Akiyama et al., 2005; Yan et al., 2005) aimed at quantifying GHG emissions and practices that affect them are unlikely to be valid for US rice systems, which differ inherently from those typically found in Asia. Differences include (i) a single annual rice crop with the exception of ratoon cropping, explained below, (ii) distinct growing and winter seasons, (iii) that rice is direct seeded rather than transplanted, (iv) improved water management due to laser leveling and reliable water supply, (v) a greater degree of mechanization, (vi) higher yield potential, and (vii) different rice cultivars (typically high-yielding temperate and tropical *japonica* cultivars). Ratoon cropping is the practice of harvesting the main crop and then allowing an additional harvestable crop to grow from the remaining stubble. Ratoon cropping is limited but occurs in the southernmost areas of the mid-South where the longer growing season permits (i.e., Gulf Coast of Texas and Louisiana).

Given these differences, a quantitative review of GHG emissions from US rice systems is needed. Until recently, there have not been sufficient data to quantify emissions from US rice systems and evaluate effects of major practices over large regional scales. Prior to 2010, much of the GHG research in the United States was limited to that done in the 1990s; however, since 2010, there has been a relatively large body of research conducted. Brye et al. (2016) identified some of the major factors affecting $CH_4$ emissions from rice fields in Arkansas; however, their data were limited to four site-years. On the basis of all suitable GHG emissions data from US rice fields, our objectives were (i) to quantify and estimate average $CH_4$ and $N_2O$ emissions for major rice growing regions within the United States, and (ii) to test the hypothesis that agronomic management practices and soils influence emissions and to quantify these effects. These findings can provide guidance toward the development of a Tier-2 approach for US rice systems. Although findings will be most appropriate to US rice systems, they will also be useful for direct-seeded systems in other temperate regions of the world, such as Australia, Europe, and parts of South America and Asia. The focus of this analysis was on $CH_4$ growing season emissions due to the prevalence of such data. However, $CH_4$ emissions from ratoon cropping and the fallow season, and $N_2O$ emissions for both the growing season and fallow season, were also considered.

# Materials and Methods

## Data Collection

An exhaustive literature survey of peer-reviewed publications was performed using Google Scholar for articles published before July 2017. Studies needed to meet the criteria below to be included in this analysis. First, the experiments had to be conducted in the United States. Second, $CH_4$ fluxes must have been measured under field conditions for at least the entire flooded portion of the growing season. Finally, seasonal fluxes had to be reported or easily extracted from figures or tables. In total, the literature search led to 33 studies from four states (Arkansas, California, Louisiana, and Texas), with study years ranging from 1980 to 2014 (Table 1). In addition to collecting growing season $CH_4$ emissions, other emissions data (i.e., $CH_4$ emissions from ratoon cropping and the fallow season, and $N_2O$ emissions for both the growing and fallow seasons) and available ancillary data were collected. Ancillary data included soil characteristics (i.e., soil series, clay content, pH, and C content), management practices (i.e., straw, fallow, rotation, N fertilizer inputs, and water), and rice cultivar (i.e., tall pure-line, semidwarf pure-line, and hybrid; henceforth referred to as tall, semidwarf, and hybrid, respectively).

## Estimation of Average Emissions

There are two main regions in the United States where rice is grown: the mid-South (including Texas, Arkansas, Louisiana, Mississippi, and Missouri) and California. These regions have distinct agronomic practices that are known to influence GHG emissions; therefore, average emissions were estimated separately for each region using only observations from peer-reviewed publications that used the standard agronomic practices for each region (Table 2). Standard practices were recognized for both the growing and winter fallow seasons and were considered the primary management rice production practices used within each region.

Average $CH_4$ and $N_2O$ emissions for the growing and winter fallow seasons were calculated separately. In addition, average $CH_4$ emissions ($N_2O$ not available) for ratoon rice were estimated. Average emissions were calculated as follows. After standard seasonal emissions were tabulated, outliers were removed. Outliers were observations that were ±5 SD from the weighted mean. This was done to remove data points that were extreme and would potentially have an undue influence on results. The cutoff

**Table 1. Summary of studies used for the analysis. Arkansas (AR), Louisiana (LA) and Texas (TX) were grouped into the mid-South region, whereas California (CA) was a separate region.**

| Reference | State | Study year(s) | Gas(es) examined | Soil series | Used for mean emission estimate† | Management practices examined and used in this analysis |
|---|---|---|---|---|---|---|
| Adviento-Borbe et al. (2013) | CA, AR | 2010, 2011 | $CH_4$, $N_2O$ | Various | F | |
| Adviento-Borbe and Linquist (2016) | CA | 2012 | $CH_4$, $N_2O$ | Various | G | |
| Bilek et al. (1999) | TX | 1995 | $CH_4$ | Bernard–Morey | G | Cultivar |
| Bossio et al. (1999) | CA | 1997 | $CH_4$ | Willows clay | | Crop residue management |
| Byrd et al. (2000) | TX | 1995, 1996 | $CH_4$ | Bernard–Morey | G | Cultivar |
| Ding et al. (1999) | TX | 1993 | $CH_4$ | Lake Charles clay | G | Cultivar |
| Fitzgerald et al. (2000) | CA | 1995, 1996 | $CH_4$ | Willows silty clay | G, F | Crop residue management |
| Kongchum et al. (2006) | LA | 2003 | $CH_4$ | Crowley silt loam | | AWD‡ |
| LaHue et al. (2016) | CA | 2013, 2014 | $CH_4$, $N_2O$ | Esquon–Neerdobe complex | G,F | AWD, seeding method |
| Lauren et al. (1994) | CA | 1992 | $CH_4$ | Nueva loam | | Crop residue management |
| Lindau and Bollich (1993) | LA | 1991 | $CH_4$ | Crowley silt loam | G | |
| Lindau et al. (1991) | LA | 1990 | $CH_4$ | Crowley silt loam | G | |
| Lindau et al. (1993) | LA | 1991 | $CH_4$ | Crowley silt loam | G | Sulfur |
| Lindau et al. (1994) | LA | 1992 | $CH_4$ | Crowley silt loam | | Sulfur |
| Lindau et al. (1995) | LA | 1993 | $CH_4$ | Crowley silt loam | G, R | Cultivar |
| Lindau et al. (1998) | LA | | $CH_4$ | Crowley silt loam | G | Sulfur |
| Lindau (1994) | LA | 1992 | $CH_4$ | Crowley silt loam | G | Sulfur |
| Linquist et al. (2015) | AR | 2012, 2013 | $CH_4$, $N_2O$ | Dewitt silt loam | | AWD, crop residue management |
| McMillan et al. (2007) | CA | 2002 | $CH_4$ | Willows clay | G, F | |
| Pittelkow et al. (2013) | CA | 2010, 2011 | $CH_4$, $N_2O$ | Clear lake clay | G, F | |
| Pittelkow et al. (2014) | CA | 2008 | $CH_4$ | Esquon–Neerdobe complex | G | Seeding method |
| Redeker et al. (2000) | CA | 1998, 1999 | $CH_4$ | Willows clay | G | Crop residue management |
| Rogers et al. (2014a) | AR | 2011 | $CH_4$ | Dewitt silt loam | G | |
| Rogers et al. (2014b) | AR | 2012 | $CH_4$ | Dewitt silt loam | G | Crop residue management, cultivar |
| Rogers et al. (2017) | AR | 2013 | $CH_4$ | Dewitt silt loam, Sharkey clay | | Sulfur, crop residue management |
| Sass et al. (1992) | TX | 1991 | $CH_4$ | Bernard–Morey | | AWD |
| Sass et al. (1994) | TX | 1991, 1992 | $CH_4$ | Lake Charles clay, Bernard–Morey | | Crop residue management |
| Sass et al. (2002) | TX | 2000 | $CH_4$ | Edna fine sandy loam | G | |
| Sigren et al. (1997a) | TX | 1994, 1995 | $CH_4$ | Bernard–Morey | G | Cultivar |
| Sigren et al. (1997b) | TX | 1994, 1995 | $CH_4$ | Bernard–Morey, mixed Bernard–Edna | G | AWD |
| Simmonds et al. (2015a) | CA, AR | 2011, 2012 | $CH_4$, $N_2O$ | Various | G | Cultivar |
| Smartt et al. (2016) | AR | 2013 | $CH_4$ | Sharkey clay | G | Crop residue management, cultivar |
| Smith et al. (1982) | LA | 1980 | $N_2O$ | Crowley silt loam | G | |
| Yao et al. (2001) | TX | 1997 | $CH_4$ | Bernard–Morey | G | |

† Letters in this column indicate if at least one observation within these studies was used to estimate average greenhouse gas emissions during the growing season (G), ratoon crop (R), or winter fallow season (F).

‡ AWD, alternate wetting and drying.

**Table 2. Summary of the characteristics assigned as the "standard" practice for estimation of greenhouse gas emissions for each region for both the growing and winter-fallow seasons.**

| Practice | Mid-South | California |
|---|---|---|
| | **Growing season** | |
| Crop rotation | Rotated with soybean | Continuous rice |
| Previous crop rice straw management | Not applicable—previous crop not rice | Incorporated after harvest |
| Previous winter water management | As rainfall dictates | Flooded |
| Seeding method | Drill seeded (continuously flooded from the 3–6 leaf stage to final drain for harvest) | Water seeded (continuously flooded from seeding to final drain before harvest) |
| Cultivar | Semidwarf, nonspecialty, nonhybrid, long-grain cultivars | Semidwarf, nonspecialty, nonhybrid, medium-grain cultivars |
| Nitrogen fertilizer | N-fertilized (if N-rate study, most appropriate rate was used) | N-fertilized (if N-rate study, most appropriate rate was used) |
| Green manure or farmyard manure | None | None |
| Sulfate additions | None | None |
| Ratoon | None | None |
| | **Winter fallow season** | |
| Fertilizer N in previous rice crop | Must have had N fertilizer applied | Must have had N fertilizer applied |
| Winter straw management | Straw retained in field | Straw retained in field |
| Winter water management | Not intentionally flooded | Flooded |

value of ±5 SD is conservative and has been used in other meta-analytic studies (Pittelkow et al., 2015; Carrijo et al., 2017). For the estimation of average emissions, no data were considered outliers. Observations were then weighted according to the number of replicates and the number of observations in each dataset from the same year with the same soil series. This was done to limit the bias from observations conducted on the same soil series in the same year as follows:

$$\text{Weight} = n_{\text{rep}} / n_{\text{obs}} \quad [1]$$

where $n_{\text{rep}}$ is the number of experimental replicates, and $n_{\text{obs}}$ is the number of $CH_4$ emission observations from the same soil series in the same year. To prevent extraordinarily large weights from studies with many experimental replicates, the number of replicates that could contribute to the weighting was limited to four. Two studies had observations with more than four replicates: McMillan et al. (2007) had six replicates and Sass et al. (2002) had 24 replicates. The weighted mean was then calculated and considered the average $CH_4$ emission for the region. Finally, bootstrapped 95% confidence intervals (CIs) were generated using the "boot" package in R with 4999 iterations. The R statistical software (R Core Team, 2016) was used for data analysis.

Methane emissions are presented as seasonal emissions with units of kilograms of $CH_4$ per hectare per season. Daily emissions, used by the IPCC, were also quantified for studies where the number of cropping days was provided or could be estimated. In this case, the seasonal emissions for each study were divided by the number of cropping days in that study. Similar to the seasonal emissions, observations were weighted (Eq. [1]) to determine the average daily emissions for each region.

There was considerable variation in reported seasonal $CH_4$ emissions within each region. To explore the causes of variation, the effects of study year and soil properties (i.e., pH, C, and clay content) were examined using backward elimination stepwise regression analysis (Hocking, 1976). Specifically, a full model with soil pH, soil C, soil clay content, and study year was developed for each region:

$$CH_4 = a + B_1(\text{pH}) + B_2(\text{C}) + B_3(\text{clay}) + B_4(\text{year}) + e \quad [2]$$

where $CH_4$, pH, C, clay, and year refer to the $CH_4$ emissions (kg $CH_4$ ha$^{-1}$ season$^{-1}$), soil pH, soil C (g C kg soil$^{-1}$), soil clay content (%), and study year, respectively, for each observation. Coefficient $a$ corresponds to the intercept for the model, whereas $e$ corresponds to the error associated with the model. The terms $B_1$, $B_2$, $B_3$, and $B_4$ correspond to the coefficients for each included parameter. Then, the least significant term (i.e., the term with the largest $P$-value) was sequentially removed and the model was reassessed until only significant ($P < 0.05$) terms remained (Supplemental Tables S1–S8).

## Management Factors Affecting Growing Season Methane Emissions

A number of management practices were evaluated for their effect on growing season $CH_4$ emissions. In selecting relevant practices to evaluate, consideration was given to those practices identified by Brye et al. (2016) (crop rotation and cultivar) and other practices with a known mechanistic cause for affecting emissions. In both regions, alternate wetting and drying (AWD), previous crop, rice straw burning, and S additions (sulfate-containing fertilizers and amendments) were evaluated. Additionally, in California, the effects of seeding method and winter flooding were evaluated, whereas the effect of cultivar type was evaluated for the mid-South.

For most practices, a meta-analytic approach was used to analyze the effect of various management practices on $CH_4$ emissions. Only peer-reviewed publications, which included side-by-side comparisons of two identical practices, except for the management practice in question, were used. Similar to other quantitative reviews and meta-analyses (Linquist et al., 2012a; Carrijo et al., 2017), the natural logarithm of the response ratio ($CH_4$ emissions in units of kg $CH_4$ ha$^{-1}$ season$^{-1}$) was used as the effect size:

$$\text{Effect size} = \ln\left[\frac{CH_4 \text{ emissions with practice}}{CH_4 \text{ emissions without practice (control)}}\right] \quad [3]$$

Effect sizes were weighted in the same manner as the average emissions observations (Eq. [1]). Two outliers (observations ±5 SD from the mean of the weighted effect sizes) were removed—one from the "AWD multiple drain" dataset, and another from the sulfur dataset. Finally, the mean effect size of each factor was calculated as the mean of the weighted effect sizes of the observations, and bootstrapped 95% CIs were generated using the "boot" package in R with 4999 iterations. The mean effect size of each practice was considered significantly different from the control if its CI did not overlap zero. For ease of interpretation, the back-transformed effect sizes are presented as the percentage change caused by each management practice in relation to the control. This value can also be converted into a scaling factor for an IPCC Tier-2 methodology using one of the two equations below:

For reductions in $CH_4$,

$$\text{Scaling factor} = 1 - \left(\frac{\%\text{ effect on } CH_4 \text{ emissions}}{100}\right)$$

For increasing $CH_4$,

$$\text{Scaling factor} = 1 + \left(\frac{\%\text{ effect on } CH_4 \text{ emissions}}{100}\right)$$

The focus of this analysis was on the percentage change caused by each management practice in relation to the control.

For S additions, the relationship between the amount of S added and $CH_4$ emissions appeared to be linear until a point at which it plateaued. Therefore, a piecewise regression approach was used to identify the S rate at which it plateaued and the slope of the two linear regression lines on both sides of that point (Toms and Lesperance, 2003). Specifically, breakpoint analysis was conducted to determine the convergence point of the two linear regression lines seeking to minimize overall deviance. Additionally, the regression equation below the threshold S rate was forced through the origin.

### Accessing the Effects of Multiple Practices

To determine an adjusted $CH_4$ emission factor, IPCC (Eggleston et al., 2006) methodology multiplies the baseline emission factor by scaling factors appropriate to various management practices. The methodology allows for the use of more than one scaling factor. The assumption here is that the effect of these practices on $CH_4$ emissions are not correlated with each other and thus, when combined, will not result in any synergistic or antagonistic effects on the resulting $CH_4$ emission estimates. To our knowledge, the practice of using more than one scaling factor at a time to quantify changes in GHG emissions has not been tested. Due to time and cost restrictions involved in conducting GHG studies, few studies have evaluated the effects of mitigation practices, both alone and in conjunction with other practices. Therefore, to assess the impact of combining multiple scaling factors on the reliability of changes to $CH_4$ emissions, studies that implemented one or more of the management practices previously described were analyzed. To generate predicted emissions for each study, the appropriate scaling factors based on the effect of that management practice on $CH_4$ emissions were applied to the control (i.e., the treatment that was similar in all aspects other than the management practices in question) of the same study as follows:

$$\text{Predicted emissions} = \text{Emissions from control} \times SF_1 \times SF_2$$

where $SF_1$ and $SF_2$ refer to two separate scaling factors generated from the study in question. This predicted emissions estimate (kg $CH_4$ ha$^{-1}$ season$^{-1}$) was then compared with the observed $CH_4$ emissions (kg $CH_4$ ha$^{-1}$ season$^{-1}$) from that study. This dataset had 41 observations with one scaling factor and six observations with two scaling factors.

## Results and Discussion

### Data, Spatial, and Temporal Spread

For the estimation of mean $CH_4$ emissions under standard conditions (Table 2), 27 observations from 17 studies and 13 observations from seven studies were used for the mid-South and California, respectively (Table 1). Spatially, these observations were well distributed across the major rice-producing areas of each region (Fig. 1). However, most studies in the mid-South were conducted at formal research stations, whereas most of the studies in California were conducted at commercial rice fields.

In the mid-South, all of the Texas and Louisiana studies were conducted between 1990 and 2000, and all of the Arkansas studies were conducted after 2011. In contrast, 23 of the California observations were made after 2010, with only four observations between 1995 and 2002. Using a backward stepwise regression analysis, which accounted for study period, soil C, soil pH, and soil clay content, differences in $CH_4$ emissions were explained by the soil variables rather than the specific study period (Supplemental Fig. S1–S3, Supplemental Tables S1–S8), which is discussed below.

Data availability to calculate average $CH_4$ emission estimates for the ratoon crop, winter fallow, and $N_2O$ emissions were much more limited in number of studies, as well as being limited both temporally and spatially (Table 1). Due to these limitations, average emissions were estimated, but the effects of management practices on these emissions were not quantified.

### Average Seasonal Emissions

#### Methane Emissions: Growing Season

The average seasonal $CH_4$ emissions using standard practices (Table 2) for the mid-South were 194 kg $CH_4$ ha$^{-1}$ season$^{-1}$ and for California were 218 kg $CH_4$ ha$^{-1}$ season$^{-1}$ (Table 3). The fact that these average seasonal emissions are similar is likely coincidental, as there are a number of practices or conditions within each region that would be expected to generate either higher or lower emissions. For example, studies have shown that continuous rice systems as opposed to rotated systems (Rogers et al., 2014b; Linquist et al., 2015), and water-seeded as opposed to drill-seeded rice systems (Pittelkow et al., 2013), result in higher $CH_4$ emissions. Both continuous rice and water seeding are standard practices in California, which would suggest that California emissions would be greater than in the mid-South. In contrast, California rice soils tend to have more clay (46%) than those in the mid-South (23%) (Table 4, Fig. 2). Soils with higher clay content tend to have lower $CH_4$ emissions (Wang et al., 1993; Baldock and Skjemstad, 2000).

The IPCC (Eggleston et al., 2006) used a baseline emission factor of 1.3 kg $CH_4$ ha$^{-1}$ d$^{-1}$ (ranging from 0.8–2.2 kg $CH_4$ ha$^{-1}$ d$^{-1}$), per Yan et al. (2005). The number of cropping days per season (i.e., planting to harvest) averaged 133 and 140 d for the mid-South (14 observations from 12 studies) and California (13 observations from seven studies), respectively. Based on these studies, the average daily emissions were 1.9 (CI: 0.98–2.71) and 1.6 (CI: 1.01–2.24) kg $CH_4$ ha$^{-1}$ d$^{-1}$ for the mid-South and California, respectively (data not shown), which is slightly higher than the mean IPCC (Eggleston et al., 2006) estimate, but well within the error range.

Given the large variability in emissions within each region, a backward step-wise regression analysis was performed to assess the effects of study year, clay content, soil C, and pH (soil property data for each region, Table 4) to determine which factors affected seasonal $CH_4$ emissions. According to this analysis, in both regions, only clay content was significant ($P < 0.05$, Fig. 2, Supplemental Tables S1–S8). Clay explained 25 to 41% of the variability in $CH_4$ emissions (Fig. 2). The relationship shows that for every 1% increase in clay content, $CH_4$ emissions declined by 6.1 and 8.1 kg $CH_4$ ha$^{-1}$ season$^{-1}$ in the mid-South and California, respectively. Higher clay contents are known to adsorb and protect soil organic C from decomposition (Baldock and Skjemstad, 2000), which decreases the methanogenic source strength of the soil. In addition, clayey soils are known to entrap $CH_4$, limiting $CH_4$ transport from soil to the atmosphere due to the low gas diffusivity (Wang et al., 1993). Yan et al. (2005) reported that both soil C and pH influenced $CH_4$ emissions. The lack of response to these soil variables in this analysis may be due to the relatively limited range of each soil variable represented by these studies. Soil C in both regions ranged from <1 to 18 g C kg$^{-1}$, and soil pH was acidic, with pH values ranging from 4.8 to 6.6, with the exception of one location (in the mid-South) that had a soil pH of 7.1 (Table 4). In contrast, in the Yan et al. (2005) study, soil C ranged from 5 to 60 g C kg$^{-1}$ and soil pH ranged from 4.5 to 8.

#### Methane Emissions: Ratoon

Methane emissions from ratoon cropping were determined from a single study (Lindau et al., 1995) and averaged 540 kg

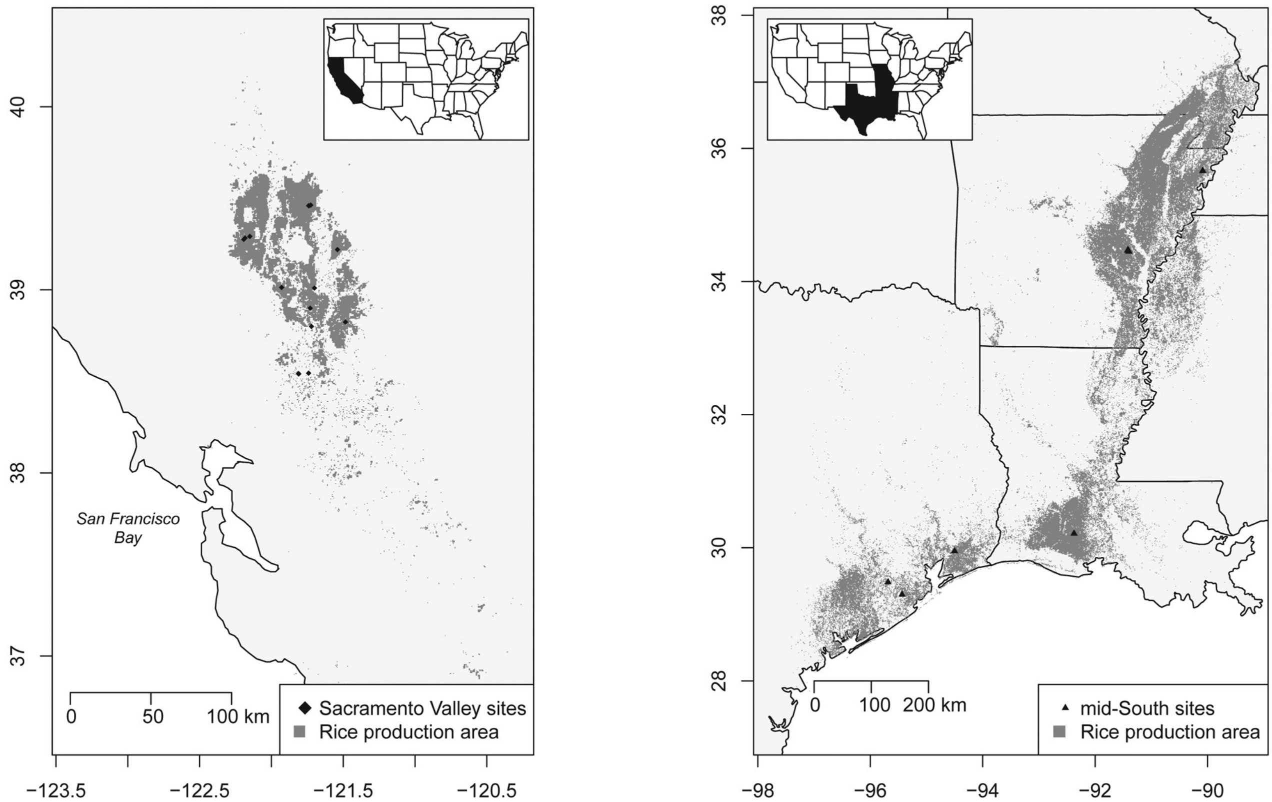


**Fig. 1. Study site locations for methane emissions (background map source: USDA-NASS, 2016; rice growing area is shown shaded). The figure on the left shows the rice growing areas in California (mostly in the Sacramento Valley), whereas the one on the right shows the rice growing areas in the mid-South.**

**Table 3. Average seasonal $CH_4$ and $N_2O$ emissions for each region for growing and fallow seasons. Data (except when noted) are all from observations using the standard practice for that region (Table 2). Lower and upper limits represent bootstrapped 95% confidence levels for the mean. Minimum and maximum values, as well as studies and observations, are also reported.**

| Region | Studies | Observations | Weighted mean | Lower limit | Upper limit | Min. | Max. |
|---|---|---|---|---|---|---|---|
| | | | kg ha$^{-1}$ season$^{-1}$ | | | | |
| | | | | | Growing season ($CH_4$) | | |
| Mid-South (main crop) | 17 | 27 | 194 | 129 | 260 | 9 | 510 |
| Mid-South (ratoon) | 1 | 3 | 540 | 465 | 614 | 468 | 629 |
| California | 7 | 13 | 218 | 153 | 284 | 67 | 446 |
| | | | | | Growing season ($N_2O$) | | |
| Mid-South (main crop) | 3 | 4 | 0.13 | 0.07 | 0.18 | 0.06 | 0.17 |
| Mid-South (ratoon) | na† | na | na | na | na | na | na |
| California | 3 | 8 | 0.15 | −0.09 | 0.39 | −0.17 | 0.66 |
| | | | | | Fallow season ($CH_4$) | | |
| Mid-South | 1 | 3 | 0.63 | 0.25 | 1.02 | 0.24 | 1.08 |
| California | 5 | 23 | 79 | 50 | 110 | 10 | 215 |
| | | | | | Fallow season ($N_2O$) | | |
| Mid-South | 1 | 3 | 1.96 | 1.53 | 2.39 | 1.47 | 2.41 |
| California | 3 | 18 | 0.65 | 0.39 | 0.90 | 0.20 | 2.24 |

† na, not available.

**Table 4. Soil percentage clay, pH, and C, separated by region, from the studies used in this analysis showing mean and range (minimum and maximum) values. Not all studies reported these values, but the number that did report is in parentheses following the mean.**

| Region | Clay | | Soil pH | | Soil C | |
|---|---|---|---|---|---|---|
| | Mean (no.) | Range | Mean (no.) | Range | Mean (no.) | Range |
| | % | | | | g kg$^{-1}$ | |
| Mid-South | 23 (23) | 12–57 | 6.1 (11) | 5.8–7.1 | 6.28 (18) | 1.11–14.8 |
| California | 46 (11) | 28–57 | 5.6 (11) | 4.8–6.6 | 11.17 (11) | 1.74–17.5 |

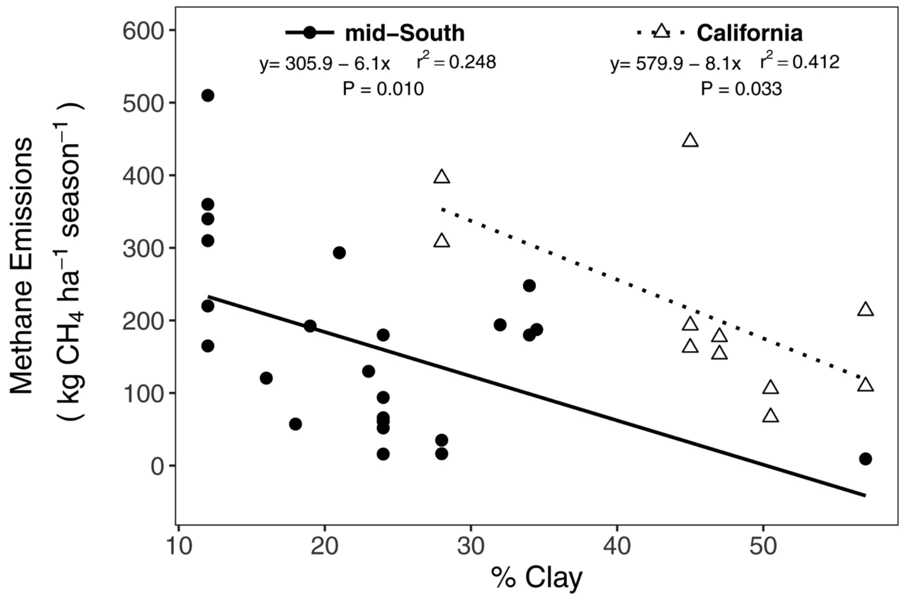


**Fig. 2. The relationship between soil percentage clay and $CH_4$ emissions for the mid-South and California. Only emissions from standard practice (Table 2) observations were included in the analysis.**

$CH_4$ $ha^{-1}$ $season^{-1}$, which is 2.8 times greater than average main crop emissions of all studies in this analysis (Table 3). Similarly, in the Lindau et al. (1995) study, the ratoon crop had 2.9 times greater $CH_4$ emissions than the main crop of that study. High ratoon $CH_4$ emissions are likely the result of: (i) all of the straw and labile C from the main crop being left in field, (ii) the field being flooded again for the ratoon crop, (iii) the crop being fertilized with N fertilizer, which promotes decomposition, and (iv) the ratoon crop typically beginning in August, when soil, water, and air temperatures are still warm. In the two studies that evaluated ratoon $CH_4$ emissions in the United States (Lindau and Bollich, 1993; Lindau et al., 1995), both reported that the amount of straw residue at the start of the ratoon had a significant effect on $CH_4$ emissions during the ratoon crop. Higher amounts of straw residues due to either the use of tall cultivars (Lindau et al., 1995) or added straw (Lindau and Bollich, 1993) led to increased $CH_4$ emissions. It should be noted that this emission estimate is based on only three observations from a single study (Lindau et al., 1995; Table 1), conducted in the 1990s at one location in Louisiana on a low clay (12%) Crowley (fine, smectitic, thermic Typic Albaqualfs) silt loam. Lindau and Bollich (1993) also evaluated $CH_4$ emissions from a ratoon crop, but this study was not included in the current analysis because they added straw to the plots in excess of what would be typical. Due to the limited data available, further research is needed to improve estimates of $CH_4$ emissions from ratoon cropping across management factors, locations, and soil types.

### Nitrous Oxide Emissions: Growing Season

There are relatively few observations of seasonal $N_2O$ emissions from standard rice production practices in the United States (i.e., 6 studies with 12 observations) (Table 1). Although $N_2O$ emissions were variable (especially in California), there was no difference in average $N_2O$ emissions between the mid-South and California, which averaged 0.14 kg $N_2O$ $ha^{-1}$ $season^{-1}$ (Fig. 3A). Converting this value to units of $N_2O$-N (dividing by 1.57), which the IPCC (Eggleston et al., 2006) used, gave 0.09 kg $N_2O$-N $ha^{-1}$ $season^{-1}$.

On the basis of a review of mostly Asian rice fields by Akiyama et al. (2005), the IPCC (Eggleston et al., 2006) determined the default $N_2O$ emissions from continuously flooded rice fields to be 0.003 kg $N_2O$-N $kg^{-1}$ N applied. Assuming a typical fertilizer N rate of 170 kg N $ha^{-1}$ for US rice fields (Linquist et al., 2009), this would amount to 0.51 kg $N_2O$-N $ha^{-1}$ $season^{-1}$, which is six times higher than reported here. Lower $N_2O$ emissions in US rice systems relative to Asian rice systems may be due to better water control, which ensures that fields remain flooded when desired. Both nitrification and denitrification processes can produce $N_2O$ (Klemedtsson et al., 1988); however, maintaining a flood keeps the soil in an anaerobic state, which limits nitrification and denitrification of $NH_4$–based fertilizers (Buresh et al., 2008), thus reducing potential $N_2O$ losses. Furthermore, a large portion of the $N_2O$ that is produced in anoxic, submerged rice soils is further reduced and emitted as $N_2$ (Firestone and Davidson, 1989; Hou et al., 2000; Aulakh et al., 2001a), which has no impact on atmospheric GHG levels. Importantly, when water is managed to allow the soil to become aerobic (e.g., for AWD management), $N_2O$ emissions may increase.

In the few studies that examined N application rates on $N_2O$ emissions, there was no relationship between N rate and $N_2O$ emissions, with $N_2O$ emissions remaining low except when rates were applied above optimal N rates ($\geq$200 kg N $ha^{-1}$, Fig. 3B; Linquist et al., 2009). These results are similar to those of Pittelkow et al. (2014), who reported that when fertilizer N was applied at optimal or suboptimal rates to rice crops, especially those that stay continuously flooded, $N_2O$ emissions were low and had little relationship to the amount of N applied. However, Pittelkow et al. (2014) also reported that when fertilizer N was applied at greater than optimal rates, $N_2O$ emissions increased rapidly.

### Winter Fallow Season Methane and Nitrous Oxide Emissions

There are few studies reporting winter fallow GHG emissions using standard winter fallow management practices (Table 2).

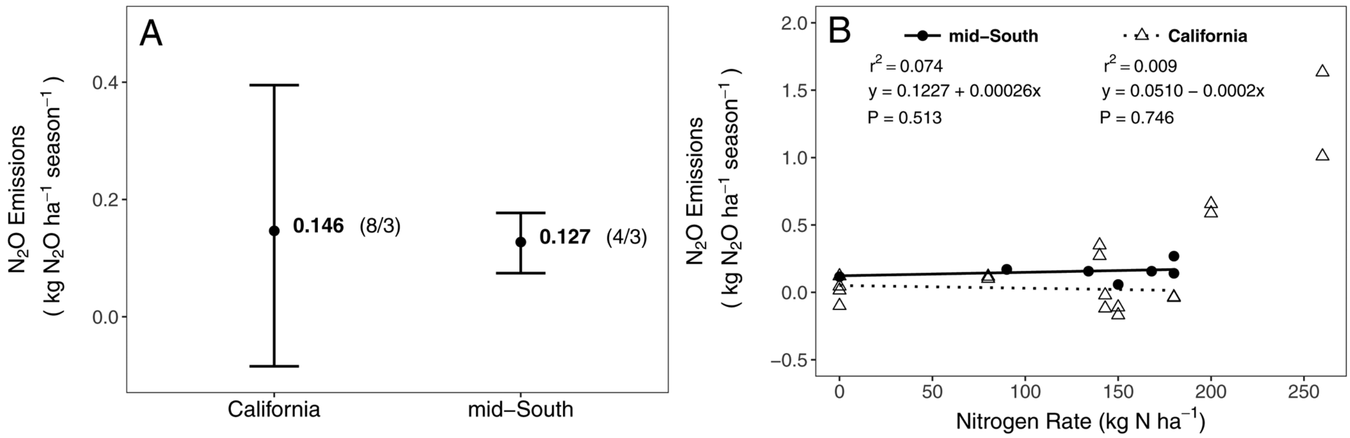


**Fig. 3. (A) Seasonal $N_2O$ emissions (kg $N_2O$ ha$^{-1}$ season$^{-1}$) by region and (B) the relationship between fertilizer N rate and $N_2O$ emissions. The linear relationship shown between N rate and fertilizer is only for rates <200 kg ha$^{-1}$. Error bars represent bootstrapped 95% confidence intervals. In panel A, numbers in parentheses refer to the number of observations/number of studies used to quantify average emissions.**

In the mid-South, only one study (three total observations) has reported on winter fallow $CH_4$ and $N_2O$ emissions. In California, five studies (23 total observations) and three studies (18 total observations) have reported on $CH_4$ and $N_2O$ emissions, respectively. In the mid-South, winter fallow $CH_4$ emissions averaged 0.63 kg $CH_4$ ha$^{-1}$ season$^{-1}$; in contrast, $CH_4$ emissions in California were higher and averaged 79 kg $CH_4$ ha$^{-1}$ season$^{-1}$, or 27% of total annual season $CH_4$ emissions (Table 3). Low winter $CH_4$ emissions in the mid-South were the result of the rice fields not being intentionally flooded during the winter. In contrast, study fields in California were intentionally flooded and were potential sources of $CH_4$ emission. In both regions, farm fields are often flooded to facilitate straw decomposition (Linquist et al., 2006) and provide waterfowl habitats. Therefore, for the mid-South in particular, more research on winter fallow $CH_4$ emissions is required under different management practices.

Winter fallow $N_2O$ emissions averaged 1.96 and 0.65 kg $N_2O$ ha$^{-1}$ season$^{-1}$ in the mid-South and California, respectively (Table 3). In both regions, $N_2O$ emissions were higher during fallow than during the growing season, although this difference was much more pronounced in the mid-South than in California, likely due to the fact that fields were not flooded during the winter. Furthermore, in the mid-South, all of the fallow season $N_2O$ emissions occurred shortly after harvest following rainfall in the fall when temperatures were still relatively warm. Given the limited data, these winter fallow $N_2O$ emissions need to be interpreted with caution and likely vary due to rainfall and temperatures during the fallow period, as reported in other cropping systems in temperate regions (Wagner-Riddle et al., 2007).

## Factors Affecting Growing Season Methane Emissions

Management factors were only considered for growing season $CH_4$ emissions due to the limited number of studies examining practices that influence ratoon crop and winter fallow $CH_4$ emissions or $N_2O$ emissions in either the growing or fallow seasons. Methane is the end product of organic matter decomposition under anaerobic conditions (Conrad, 2002); therefore, water and C management are the two primary factors affecting $CH_4$ emissions. In general, the effects of management practices on GHG emissions reflect this. Management practices such as AWD and seeding method (both related to water management) and residues (C management) had the largest impact on $CH_4$ emissions, with reductions ranging from 39 to 83% (Table 5). These reductions were similar to those of Yan et al. (2005) in their review of major variables controlling $CH_4$ emissions from Asian rice systems.

### Previous Crop and Residue Management

Residues left in the field from the previous crop can increase $CH_4$ emissions the following season, as residues provide a C substrate for methanogenesis during the flooded rice-cropping season (Yan et al., 2005). The two US regions vary in the amount of residue left in the field before the standard growing season (Table 2). In California, the standard practice is to continuously plant rice year after year. Assuming a harvest index of 0.5 (Dobermann and Fairhurst, 2000), there is ∼9000 kg straw ha$^{-1}$ after rice harvest (rice straw contains roughly 50% C; www.ecn.nl/phyllis). In California, it is the standard practice to incorporate rice straw into the soil and flood the field after harvest (Table 2) to promote straw decomposition during the winter fallow period. Linquist et al. (2006) reported that roughly 50% of the straw decomposed during the fallow period, and thus there is ∼4500 kg straw ha$^{-1}$ remaining in the soil at the onset of the next growing season. In contrast, in the mid-South, there is relatively little residue returned to the soil, as the previous crop is usually soybean [*Glycine max* (L.) Merr.], which returns less plant biomass to the soil than rice. Assuming a soybean grain yield of 3300 kg ha$^{-1}$ (a rough average for Arkansas; USDA, 2017) and a harvest index of 0.43 (Brye et al., 2004), there is ∼4370 kg residue ha$^{-1}$. Furthermore, the relative amount of soybean residue remaining after winter fallow in the mid-South would likely be less than that for rice straw, as soybean residue decomposes at faster rates than cereal residues (Xu et al., 2017).

The differences in cropping systems and residue management are likely to affect the following season's $CH_4$ emissions. We evaluated the effects on $CH_4$ emissions of (i) soybean grown as the previous crop, (ii) fallow in previous growing season, and (iii) rice grown as the previous crop with the rice straw being burned versus when rice was the previous crop and the straw residue was left in the field during the winter fallow. There was no difference in the next growing season's $CH_4$ emissions between rice fields being flooded or not during the previous fallow (data not shown). For this analysis, only studies that had incorporated

Table 5. The effect of various management practices on $CH_4$ emissions and scaling factors grouped by region compared with the standard practice (Table 2). The bootstrapped 95% confidence intervals for each are provided in parenthesis.

| Region | Scaling factor | Study no. | Observation no. | Effect on $CH_4$ (relative to standard) | Scaling factor |
|---|---|---|---|---|---|
| | | | | % | |
| Mid-South | High crop residue† | 9 | 23 | 116 (72 to 174) | 2.16 (1.72 to 2.74) |
| | AWD‡ (single drain) | 4 | 9 | −39 (−47 to −30) | 0.61 (0.53 to 0.70) |
| | AWD (multiple drains) | 3 | 10 | −83 (−91 to −65) | 0.17 (0.09 to 0.35) |
| | Sulfur | 5 | 14 | Variable§ | |
| | Cultivar (CLXL745) | 3 | 8 | −26 (−37 to −12) | 0.74 (0.63 to 0.88) |
| | Cultivar (tall cultivars) | 7 | 32 | 31 (13 to 50) | 1.31 (1.13 to 1.50) |
| California | Little or no crop residue† | 9 | 23 | −54 (−63 to −42) | 0.46 (0.37 to 0.58) |
| | AWD (single drain) | 4 | 9 | −39 (−47 to −30) | 0.61 (0.53 to 0.70) |
| | AWD (multiple drains) | 3 | 10 | −83 (−91 to −65) | 0.17 (0.09 to 0.35) |
| | Sulfur | 5 | 14 | Variable§ | |
| | Seeding method (drill seeded) | 2 | 3 | −60 (−68 to −48) | 0.40 (0.32 to 0.52) |

† High crop residue refers to nonharvested plant biomass from a high-residue crop (like rice or corn [*Zea mays* L.]) being left in the field from the previous season. Little or no crop residue refers to situations in which straw was burned, the field was fallow the previous year, or a low-residue crop was grown (in this case, soybean).

‡AWD, alternate wetting and drying.

§ A linear relationship exists between amount of sulfur added and the percentage reduction in $CH_4$ emissions. For every 30 kg S $ha^{-1}$ added (up to a maximum of 338 kg S $ha^{-1}$), $CH_4$ emissions are reduced by 4%.

naturally present rice straw into the soil in the fall were included (i.e., studies that added exogenous inputs of rice straw prior to planting were not considered). All of the three aforementioned practices represent conditions with limited C input from residues from the previous season and, on average, reduced $CH_4$ emissions by 14 to 57% versus when rice was the previous crop and the straw was left in the field after harvest (Fig. 4). Since the mechanism responsible for reducing $CH_4$ emissions was similar (i.e., limited C to promote methanogens), these practices were grouped into a single practice representing limited C input from the previous year. This resulted in an average $CH_4$ reduction of 54% versus when rice was the previous crop and the straw was left in the field after harvest.

The effect of previous C inputs was assumed to be similar in the mid-South and in California for this analysis; however, due to the regions having different standard practices, the effect on standard $CH_4$ emissions was realized in opposite directions (Table 5). For example, since the standard practice in the mid-South is to have little residue returned to the soil, having rice straw from the previous season increased $CH_4$ emissions by 116%. In contrast, in California, since the standard practice is to have rice straw returned to the soil, not having rice straw from the previous season reduced $CH_4$ emissions by 54%.

### Water Management

Alternate wetting and drying is a water management practice that is known to decrease $CH_4$ emissions from rice fields (Yan et al., 2005) by introducing dry-down periods during the growing season, which create aerobic soil conditions and decrease the production of $CH_4$. The IPCC (Eggleston et al. 2006) developed scaling factors based on single or multiple dry-downs during the growing season. In this study, single and multiple dry-downs both significantly reduced $CH_4$ emissions relative to a continuously flooded control (Table 5), and multiple dry-downs

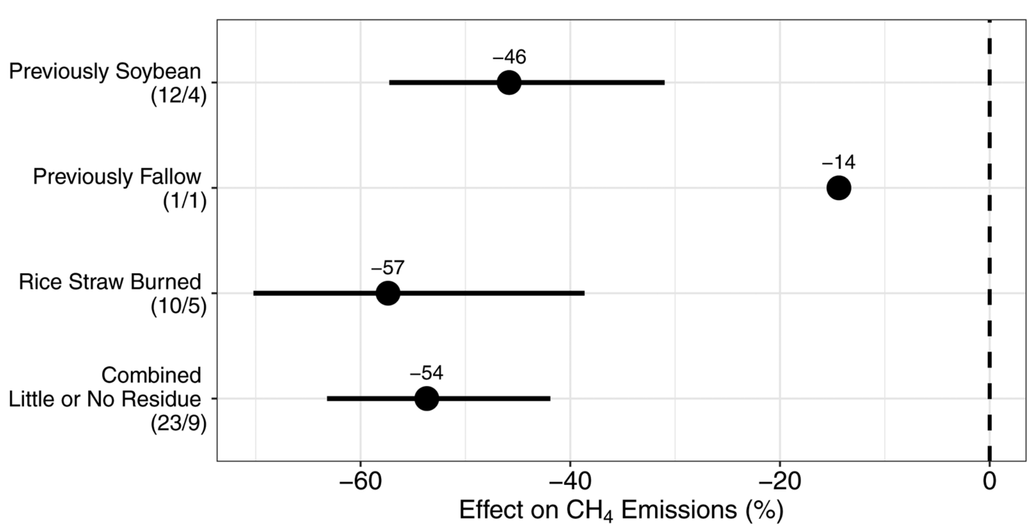


Fig. 4. The effect of postharvest residue left in the field on $CH_4$ emissions in the following season, relative to when rice was the previous crop and the residue was left in the field during fallow. The top three points show management practices that resulted in little or no crop residues in the field. The bottom point represents the effect on $CH_4$ emissions when all practices were combined for the analysis. Numbers in parentheses along the *y*-axis refer to the number of observations/number of studies used for each analysis.

reduced emissions (by 83%) significantly more than a single dry-down (by 39%). These reductions equate to scaling factors of 0.61 and 0.17 for single and multiple dry-downs, respectively. In contrast, the IPCC (Eggleston et al., 2006) reported a single drain scaling factor of 0.60 and a multiple drain scaling factor of 0.52. Although the single dry-down scaling factor is similar between the findings here and IPCC (Eggleston et al., 2006), the multiple dry-down scaling factor is much lower in this analysis (indicating a greater reduction in $CH_4$ emissions) than the IPCC (Eggleston et al., 2006). One possible reason why the IPCC methodology shows that single and multiple dry-downs are more similar in their effects on $CH_4$ emissions than in this analysis is poorly described AWD practices in the literature (i.e., to what extent the soils dry during the dry-down). It is likely that the number of days a field is drained during a dry-down period may be more important than the number of dry-downs, as Carrijo et al. (2017) reported for AWD effects on rice yields. The significant differences between single and multiple dry-downs may be because the dry-down periods in US experiments were relatively similar across experiments. Dry-down periods averaged 8.4 d, with the 25th and 75th quantiles corresponding to 6 and 10 d, respectively. Furthermore, the dry-down periods from the studies in this analysis were longer than the 3-d minimum aeration period required in the IPCC (Eggleston et al., 2006) guidelines. In addition, a given number of dry days is not necessarily a good indicator for how a drain event may affect emissions, as climate and soil type determine how fast a soil dries. Ideally, research should focus on identifying critical soil moisture levels at which desired effects (e.g., reduced $CH_4$ emissions) can be achieved. There are several possible approaches to this, including direct measurements of soil moisture content, as has been done in a number of the recent US studies (Linquist et al., 2015; LaHue et al., 2016), or measuring the subsidence of the perched water table using monitoring wells (Lampayan et al., 2015).

One concern with AWD is the potential to increase $N_2O$ emissions during the dry-down periods. Both nitrification, which can occur as the soil dries during a drain period, and denitrification, which occurs when the field is reflooded, processes can produce $N_2O$ (Klemedtsson et al., 1988). Studies that have measured both $CH_4$ and $N_2O$ have generally shown that $N_2O$ emissions increase during the dry-down, but the increase is not enough to offset the benefits from $CH_4$ reductions (Wassmann et al., 2000; Linquist et al., 2015). Only two studies have been conducted in the United States that measured $N_2O$ emissions under AWD water management, one study in California and another in Arkansas. In the California study (LaHue et al., 2016), AWD treatments resulted in seasonal $N_2O$ emissions being on average 0.015 kg $N_2O$ $ha^{-1}$ $season^{-1}$ lower than continuously flooded treatments. However, in the Arkansas study (Linquist et al., 2015), AWD treatments had, on average, 0.45 kg $N_2O$ $ha^{-1}$ $season^{-1}$ higher emissions than the continuously flooded treatments (Table 6). Low $N_2O$ emissions can be attributed to the dry-downs being conducted when soil mineral N was low. In the Californian water-seeded system, where all of the fertilizer N was applied before planting, the first dry-down occurred ~6 to 7 wk after planting, when measured soil-extractable mineral N levels were low (LaHue et al., 2016). Similarly, in the Arkansas study, the dry-down occurred ~3 wk after establishment of the permanent flood, when it is expected that soil mineral N levels would be low (Norman et al., 2013). In contrast, Lagomarsino et al. (2016) reported very high $N_2O$ emissions from AWD treatments in Italy, where dry-downs likely occurred when there were high levels of soil mineral N present, and as a result, they showed that the global warming potential ($CH_4$ + $N_2O$) was higher in the AWD treatments than in the continuously flooded treatments. This analysis highlights the importance of managing both water and N to reduce $CH_4$ and $N_2O$ emissions.

### Sulfur Management

Sulfur additions, in the form of sulfate-containing fertilizers or amendments, enhance substrate competition between sulfate-reducing bacteria and methanogens, thereby reducing $CH_4$ production and emissions in anaerobic systems (Denier van der Gon et al., 2001). For US rice systems (Fig. 5), and more broadly across all rice systems (Linquist et al., 2012a), adding up to 338 kg S $ha^{-1}$ led to linear reduction in $CH_4$ emissions, such that every 30 kg S $ha^{-1}$ added led to a 4% reduction in $CH_4$ emissions. This relationship was linear up to a threshold of 338 kg S $ha^{-1}$, which corresponded to a 45% reduction in $CH_4$ emissions versus when no S was added. Above this level, continued S additions no longer reduced $CH_4$ emissions.

Sulfur is not commonly added to rice fields in large quantities, as S deficiencies are not common in the United States. However, S is a component of a number of common fertilizer products, including ammonium sulfate (24% S), single superphosphate (14% S), and potassium sulfate (18% S). If these fertilizer products are applied at rates typically used, the mitigation effect is small due to the small amount of applied sulfate. For example, although ammonium sulfate is often used in rice systems, it is rarely used as the primary N source for rice. Instead, ammonium sulfate may be used as part of a starter N–P–K blend applied at planting or later in the season as a top-dress application. In these situations, a typical rate is 168 kg ammonium sulfate $ha^{-1}$, which corresponds to 40.5 kg S $ha^{-1}$ and, given the relationship in Fig. 5, a $CH_4$ reduction of ~5%. If ammonium sulfate was used as the primary N source and was applied at rate equivalent to 168 kg N $ha^{-1}$, this would correspond to an S application rate of 192 kg S $ha^{-1}$ and a 25% reduction in $CH_4$ emissions (i.e., a scaling factor of 0.75). In contrast, gypsum contains ~19% S and is used as a soil amendment in sodic soils, where it is typically applied in much larger quantities (e.g., 5 Mg $ha^{-1}$; Yaduvanshi and Swarup, 2005), which could have a much larger effect on $CH_4$ emissions.

### Seeding Method

In the United States, rice is sown in one of two ways—water seeding or dry seeding, which are detailed by Street and Bollich, (2003). Briefly, water seeding is the practice of flooding a field before planting rice and then sowing seed, usually by airplane, into the flood water. The field stays flooded or saturated until it is drained for harvest at the end of the season. Water seeding is the dominate practice in California, being used on ~95% of the area. Dry seeding is the practice of planting rice either in rows or broadcasting and then lightly incorporating seed into soil. The seed germinates and young seedlings are established with existing soil moisture, seasonal rainfall, and/or irrigation. At the three- to six-leaf stage (~4 wk after emergence), a permanent flood is established, which is maintained until being drained for harvest at the end of the season. Therefore, water-seeded systems are

**Table 6. Nitrous oxide emissions under continuously flooded conditions (control) compared with alternate wetting and drying (AWD) water management.**

| Reference | State | Year | Control $N_2O$ | AWD $N_2O$ | Difference |
|---|---|---|---|---|---|
| | | | kg $N_2O$ ha$^{-1}$ season$^{-1}$ | | |
| Linquist et al. (2015) | Arkansas | 2012 | 0.049 | 0.163 | 0.115 |
| | Arkansas | 2012 | 0.049 | 0.360 | 0.311 |
| | Arkansas | 2012 | 0.049 | 0.215 | 0.167 |
| | Arkansas | 2013 | 0.110 | 0.613 | 0.503 |
| | Arkansas | 2013 | 0.110 | 0.629 | 0.519 |
| | Arkansas | 2013 | 0.110 | 1.650 | 1.540 |
| | Arkansas | 2013 | −0.013 | 0.044 | 0.057 |
| | Arkansas | 2013 | −0.013 | 0.311 | 0.324 |
| | Arkansas | 2013 | −0.013 | 0.517 | 0.530 |
| | Mid-South | Mean | 0.049 | 0.500 | 0.452 |
| LaHue et al. (2016) | California | 2013 | −0.035 | −0.060 | −0.025 |
| | California | 2014 | −0.039 | −0.044 | −0.005 |
| | California | Mean | −0.037 | −0.052 | −0.015 |

flooded earlier in the season and the flood lasts ~4 wk longer than in dry-seeded systems, likely affecting $CH_4$ emissions.

Only two California studies (three total observations) have compared $CH_4$ emissions between water- and dry-seeded systems (Table 1). From these studies, dry seeding reduced $CH_4$ emissions by 60% compared with water seeding (Table 5). There are two mechanisms that probably lead to a reduction in $CH_4$ emissions under dry seeding. First, the dry-seeded fields are flooded ~1 mo later than water-seeded fields, resulting in a shorter anaerobic period. Second, in California, where rice straw is incorporated during the fallow period, ~50% of the straw remains in the fields at the start of the following season (Linquist et al., 2006). Chidthaisong and Watanabe (1997) reported that $CH_4$ emissions early in the season were largely the result of straw decomposition. Dry seeding allows the remaining residue to decompose for about a month under moist but nonflooded conditions, resulting in organic matter decomposition releasing $CO_2$ rather than $CH_4$ (Devêvre and Horwath, 2000). In the mid-South, the shorter flooded period with dry seeding would likely reduce $CH_4$ emissions relative to water seeding. However, we speculate that the reduction in $CH_4$ emissions would be less than that observed in California. This would be because, in the mid-South, rice is normally grown in a rotation with soybean that has less postharvest residue than rice (see prior discussion). The role of seeding method needs to be better quantified in mid-South rice systems.

### Cultivar

A number of studies have been conducted in the United States and elsewhere that have evaluated the relative performance of different rice cultivars on $CH_4$ emissions. Wassmann et al. (2002) concluded from nine seasons of data from Asia that differences between cultivars were inconsistent, suggesting complex interactions with the environment. These authors (Wassmann et al., 2002) stressed the need for a greater mechanistic understanding if the mitigation effect of cultivar selection is to be exploited to reduce $CH_4$ emissions. Various mechanisms have been proposed that would lead to differences in $CH_4$ emissions among cultivars, including differences in rhizospheric oxidation potential (Bilek et al., 1999; Ma et al., 2010; Jiang et al., 2017), root exudates (Aulakh, 2001b, 2001c), and the ability of the plant to transport $CH_4$

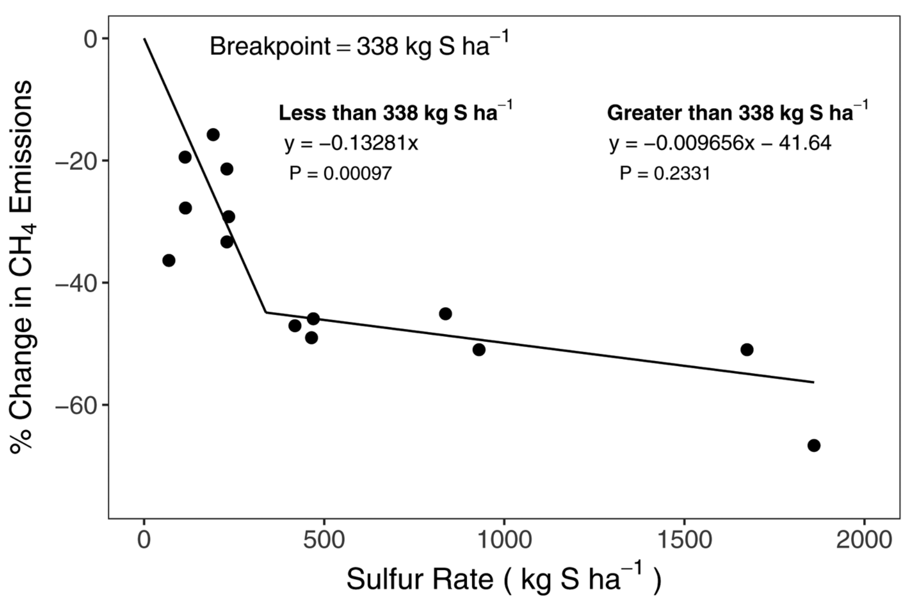


**Fig. 5. The relationship between S additions and $CH_4$ emissions using a linear plateau model. For the development of this figure, the line was forced through the intercept.**

through aerenchyma tissue that allows for gas exchange between the soil and atmosphere (Ding et al., 1999; Aulakh et al., 2000).

Results from this analysis indicate that the hybrid rice cultivar CLXL745 consistently reduced $CH_4$ emissions relative to semi-dwarf cultivars in the mid-South. This was likely due to reduced $CH_4$ emissions in the latter half of the growing season (Rogers et al., 2014b). In the latter half of the growing season, root-derived C is thought to be the main contributor to $CH_4$ emissions (Chidthaisong and Watanabe, 1997), suggesting that this hybrid has reduced leakage of photosynthetically fixed C exudates from the roots. It is possible that reduced leakage suggests greater internal efficiency of photosynthates, leading to high yields. Similarly, others have reported that high-yielding cultivars reduce seasonal $CH_4$ emissions (Huang et al., 1997; Jiang et al., 2017). Although some hybrids may result in lower $CH_4$ emissions than semidwarf cultivars (Ma et al., 2010), caution needs to be taken in assuming that this is the case for all hybrids. For example, Simmonds et al. (2015a) reported that a hybrid cultivar resulted in higher $CH_4$ emissions than when a semidwarf cultivar was planted, although this observation was only based on a single, nonreplicated study.

In general, plots planted with tall cultivars had 31% higher $CH_4$ emissions than those with semidwarf cultivars. However, most of the data from these studies were from before 1996, and these older cultivars are no longer in use. Two recent studies (Rogers et al., 2014b; Smartt et al., 2016) compared a tall cultivar (Taggart) with a semidwarf cultivar (Cheniere), and $CH_4$ emissions were not significantly different between the two cultivars in either study.

When considering specific cultivars as potential scaling factors, it should be acknowledged that rice cultivars have a relatively rapid turnover. Even good cultivars are replaced by new cultivars every 5 to 10 yr. Therefore, unless there are screening programs to evaluate emissions with each newly released cultivar, which would likely be a costly proposition, or a mechanism identified that clearly denotes whether a cultivar will result in higher or lower $CH_4$ emissions, the prospects of cultivar-specific scaling factors are slim.

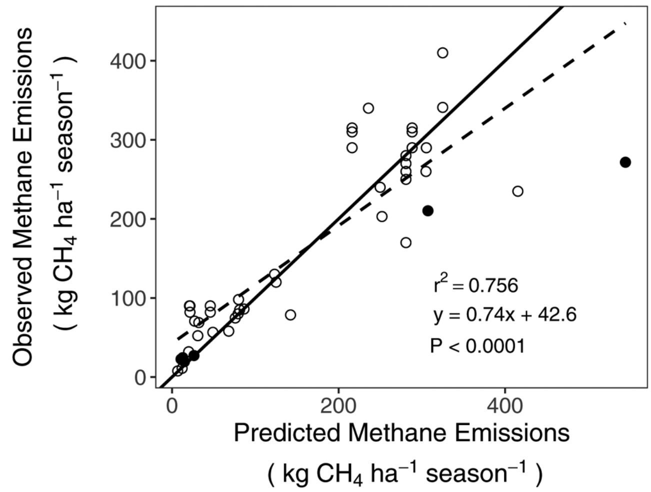


**Fig. 6. The relationship between the observed versus predicted $CH_4$ emissions when using one (open circle) or two (closed circle) practices to adjust average emissions. The solid line is the 1:1 line, and the dashed line is the best fit line through all points.**

## Use of Scaling Factors Together

Most studies evaluate the effect of a single practice on $CH_4$ emissions. Similarly, the analysis above considered the effect of only one practice at a time on emissions. However, a number of practices may be applicable and used on any given rice field. In the IPCC (Eggleston et al., 2006) methodology, users can apply scaling factors from multiple applicable practices. Doing this assumes that the effect of these practices on $CH_4$ emissions are not correlated with each other, and thus, when scaling factors are combined, there will not be synergistic or antagonistic effects on the resulting $CH_4$ emission estimates. This approach of combining scaling factors was tested using data from this study; unfortunately, this database had only six observations with which to evaluate two scaling factors. Using this limited dataset when comparing observed versus predicted $CH_4$ emissions, the

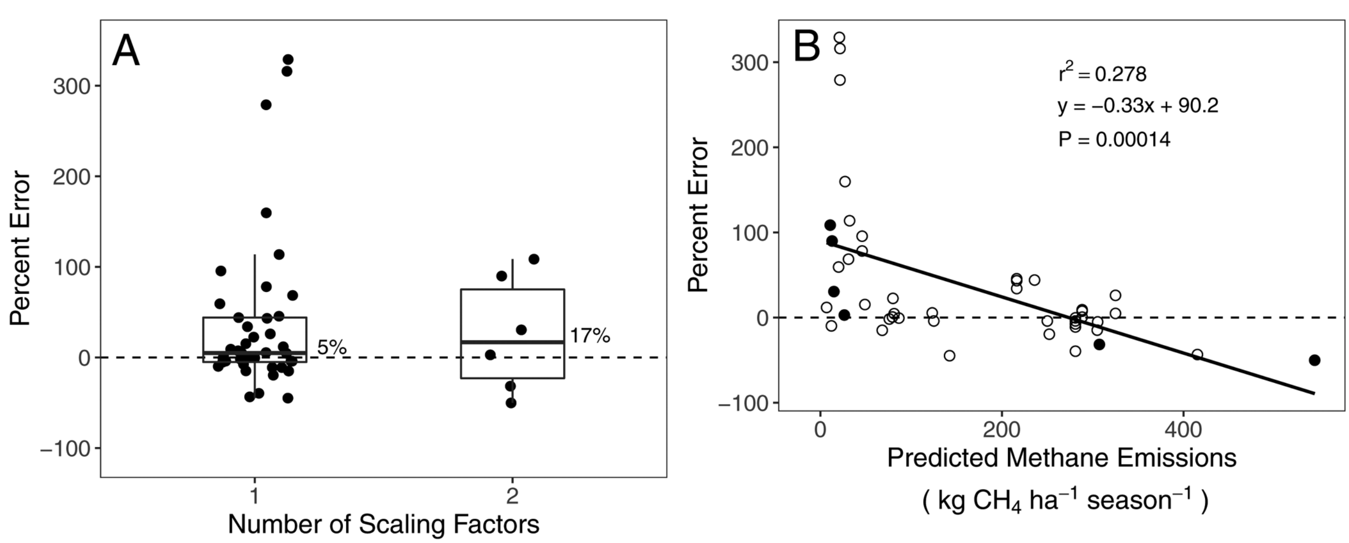


**Fig. 7. (A) The percentage error [(observed $CH_4$ – predicted $CH_4$)/predicted $CH_4$] for one and two scaling factors. Data points are staggered for visual interpretation. Points above the dashed line indicate that the observed $CH_4$ emissions were greater than the predicted $CH_4$ emissions and vice versa. A boxplot overlaying the data points indicates the median value (thick horizontal line), and the lower and upper lines of each box correspond to the first and third quartiles (the 25th and 75th percentiles). The whiskers extend from the hinge to the furthest value, but no further than 1.5 times the interquartile range (or the distance between the first and third quartiles) from the hinge. Data beyond the end of the whiskers are considered "outlying" points. (B) The percentage error [(observed $CH_4$ – predicted $CH_4$)/predicted $CH_4$] versus predicted $CH_4$ emissions.**

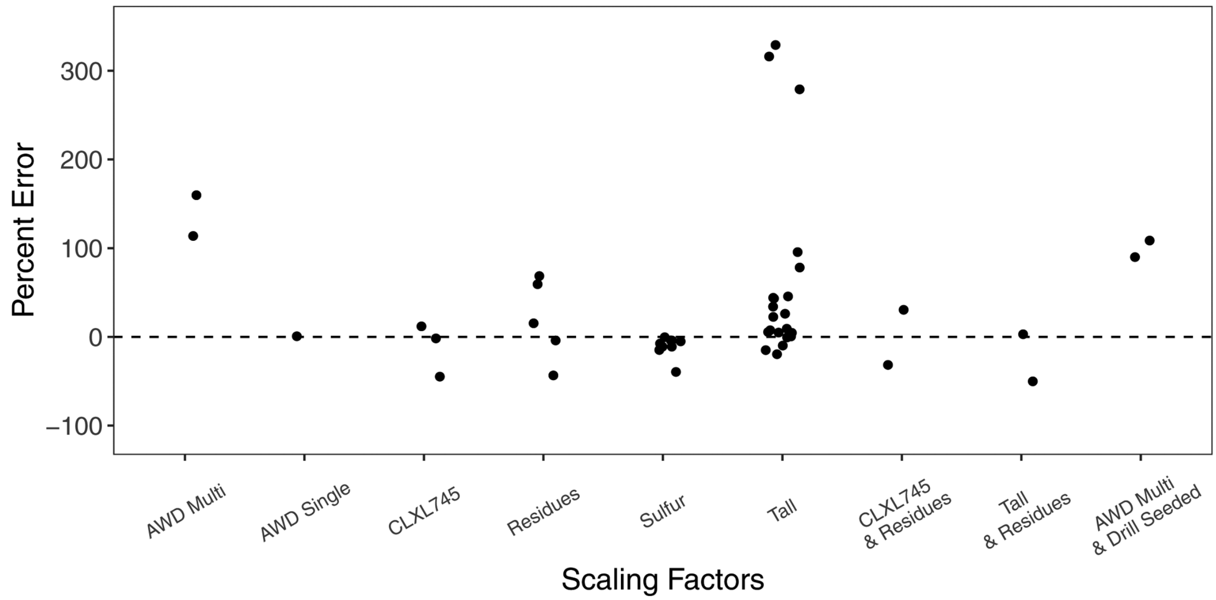


**Fig. 8. The percentage error (observed $CH_4$ – predicted $CH_4$)/predicted $CH_4$] for specific scaling factors. Data points are staggered for visual interpretation. Points above the dashed line indicate that the observed $CH_4$ emissions were greater than the predicted $CH_4$ emissions, whereas observations below the dashed line indicate that the predicted $CH_4$ emissions were greater than the observed. AWD refers to alternate wetting and drying with either multiple (multi) or single dry-down events. CLXL745 is a hybrid rice cultivar.**

majority of values fell along the 1:1 line ($r^2 = 0.76$), regardless of whether one or two practices were considered (Fig. 6).

Examining the error between predicted and observed values [(observed $CH_4$ – predicted $CH_4$)/predicted $CH_4$] using two scaling factors together did not increase the error relative to using only one scaling factor (Fig. 7a and 7b). Importantly, the largest errors were observed when predicted emissions were <50 kg $CH_4$ ha$^{-1}$ season$^{-1}$, whereas $CH_4$ emissions >50 kg $CH_4$ ha$^{-1}$ season$^{-1}$ had errors of <50% (Fig. 7b). A closer examination of the source of error indicated that three observations with errors of ∼300% were related to the use of the "tall cultivar" scaling factor by itself (Fig. 8).

## Conclusion

Average GHG emissions for US rice systems that include growing (and ratoon) and fallow seasons for both $CH_4$ and $N_2O$ were estimated using a meta-analytic approach with previously published data. In addition, the effects of a number of pertinent management practices on $CH_4$ emissions were quantified. This synthesis can be used in the development of a Tier-2 methodology for US rice systems, and possibly similar rice systems in other temperate regions of the world. This type of analysis also allows the identification of knowledge gaps, which include the following. First, ratoon rice cropping, which is practiced in many parts of the world, was found to produce higher $CH_4$ emissions than the main crop, and this practice has not been thoroughly evaluated in regard to GHG emissions. More research needs to be performed to quantify average emissions from ratoon systems, as well as to identify practices that could reduce emissions from ratoon systems. Second, both US regions practice water seeding and dry seeding; however, seeding method effects on GHG emissions have not been well studied. Third, a mechanistic understanding of varietal effects on $CH_4$ emissions needs to be developed. Fourth, a better understanding of how multiple practices affect emissions is needed. This analysis provides an initial step in this direction, but more research is required. The development of robust, process-based models and additional field observation campaigns, specifically designed to quantify the role of different management and landscape influences, is needed to improve on our current knowledge of GHG emissions from rice systems.

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## Supplemental Material

Supplemental information is provided that represents the results of specific linear regression analyses evaluating growing season $CH_4$ emissions as a function of study year, soil clay content, soil pH, and soil C. Secondly, results of a backwards step-wise regression are shown that assess which of the abovementioned factors were significant contributors to $CH_4$ emissions.

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