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Life cycle greenhouse gas emissions in California rice production

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ABSTRACT

The nexus of climate change and food security challenges currently facing humanity requires better understanding of how to balance food production needs with climate change mitigation. Life cycle assessment methods provide a way to quantify the climate impacts of a food product by accounting for all greenhouse gas (GHG) emissions associated with its production, including upstream and downstream from the farm. This study modeled life cycle GHG emissions for one kg of milled, unpackaged rice produced in California, USA, a state that achieves some of the highest rice yields in the world. The goal was to (1) provide an assessment of life cycle GHG emissions of a comparatively intensive production system, using local field emissions data, (2) identify emissions hotspots, and (3) create a model that elucidates the life cycle-wide consequences of potential changes in field management practices. Study parameters are based on an annual cropping cycle, with continuous flooding during the growing season and soil incorporation of straw post-harvest, and yields of 9.3 Mt ha^{-1} dried paddy rice. Field emissions (growing and fallow seasons) were estimated with empirical data while other emissions were calculated using an engineering model coupled with life cycle inventory datasets and vehicle emission models.

The 100-year global warming potential (GWP, based on CO_2, CH_4 and N_2O) was 1.47 kg CO_2-equivalent (CO_2e) kg^{-1} of milled rice; of which field emissions contributed 69%. These results are relatively low when compared to life cycle studies in other parts of the world, due in large part to higher grain yields and lower field emissions. When using IPCC Tier 1 estimates of field emissions, the GWP increased to 3.60 CO_2e kg^{-1} rice, highlighting the importance of using direct field measurements as we have in this study. Due to their large contributions to life cycle GWP, reducing field CH_4 emissions through different field management practices, optimizing N fertilizer use, and increasing fuel efficiency or reducing use of farm machinery present the greatest opportunities to reduce life cycle emissions. Because of high variability and uncertainty in estimating field emissions, they should also be targeted for improved measurement and modeling.

1. Introduction

Climate change and food security are increasingly recognized as two of the most pressing as well as closely interlinked challenges facing humanity in the 21st century, suggesting a need for more sustainable intensification of food production (Godfray et al., 2011). Rice is the staple crop for the largest number of people on earth (Maclean et al., 2002). However, rice production has received significant attention in the global climate change discourse due to its uniqueness among cultivated crops for emitting both methane (CH_4) and nitrous oxide (N_2O), two greenhouse gases (GHG) that are more potent than carbon dioxide (CO_2) in driving climate change. Methane emissions comprise roughly 90% of the global warming potential (GWP) of field emissions in flooded rice systems, while N_2O makes up the remainder (Linquist et al., 2012a). The US Environmental Protection Agency (US EPA, 2013) estimates that U.S. rice cultivation alone was responsible for 5.2 Tg CO_2 equivalents (CO_2e) in 2010, making it the 9th largest source of methane in the U.S. Van Groenigen et al. (2010) introduced yield-scaled GWP

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as a metric to assess sustainable intensification, with the objective of achieving the lowest emissions per unit of grain yield. Their concept only applied to field GHG emissions. An assessment of full life cycle GHG (LCGHG) emissions estimates the GWP per unit of edible product (in this case milled rice), by considering not only field emissions, but also the upstream and downstream emissions that result from decisions taken at the field level, such as use of specific materials and energy inputs. Despite the contribution of rice to regional, national, and global GHG emissions, only a few full life cycle GHG assessments of rice production have been undertaken (Blengini and Busto, 2009; Hokazono and Hayashi, 2012; Thanawong et al., 2014; Wang et al., 2010). No assessments have been done for systems with the combination of climate conditions, intensive production practices, and high grain yields, as found in California.

California produces approximately 1.8 million Mt of rice per year, about 20% of total U.S. production. California's rice yields (9.3 Mt ha^{-1}) are approximately 10% higher than national average yields (CDFA, 2013). High yields are due in part to a Mediterranean climate with high solar radiation, and a growing season characterized by little to no rainfall, low humidity, and long days. Despite lack of rainfall during the growing season, well developed irrigation schemes that make use of winter snow melt in the Sierra Nevada mountain range ensure adequate water. This yield advantage along with the effects of variety (Wassmann et al., 2002), crop management and soil type (Yan et al., 2005) in determining CH_4 and nitrous oxide (N_2O) emissions, illustrate the need for regionally-specific life cycle assessments (LCAs) of GHG emissions to understand the climate footprint of different food products.

The goals of this LCGHG assessment were (1) to develop a process-based LCGHG assessment to generate a baseline estimate of GHG emissions for production and milling of rice in the Sacramento Valley region of California that can be compared with assessments from other regions; (2) to contextualize locally measured estimates of field CH_4 and N_2O emissions within the entire rice production process and its upstream supply chain; (3) to provide information that may be used by producers, food companies, and others in the industry to target hotspots of emissions should they wish to reduce them; and (4) to assist scientists in understanding the life cycle ramifications of changes in agronomic practices that they may be studying at the field level. This study makes an important contribution to the rice LCA literature in relying on field emissions estimates for both CH_4 and N_2O from a range of field studies conducted in representative locations in the actual region of study, instead of IPCC estimates or estimates adjusted from data from other regions or systems, as is commonly done (e.g. Hokazono and Hayashi, 2012; Thanawong et al., 2014).

2. Methodology

The study described here largely conformed to internationally accepted standards for life cycle assessment (LCA) methodology as applied to GHG studies (Technical Committee ISO/TC207, 2006; British Standards Institution, 2011). Because this study was limited to assessing GHG emissions (CO_2, CH_4, and N_2O), environmental impact categories other than global warming potential (GWP) were not included.

This assessment considered processes from cultivation through fallow season field management and milling operations (Fig. 1). Packaging, retail, and home preparation were not considered in this study. Production of capital equipment (tractors, processing facilities, etc.) was also not included in this assessment, though equipment operation and the full fuel life cycle were included. Omitting production of capital equipment in agricultural LCAs is common practice and is not expected to lead to significant changes in the results, especially for GHGs (British Standards Institution, 2011; Frischknecht et al., 2007).

Table 1
Fertilizer application rates and percentage of area applied[a].

Fertilizer type	kg ha^{-1} (kg N)	% Of total area
Aqua-ammonia	672.0 (134.4)	100
Ammonium phosphate (16-20-0)	224.1 (35.9)	100
Potash	56.0 (0)	100
Zinc sulfate	16.8 (0)	50

[a] *Sources:* Mutters et al. (2007), Greer et al. (2012).

Life cycle inventory (LCI) development tracks the environmental flows associated with the system of analysis. Developing the LCI required first modeling the direct inputs and outputs from the rice production system, and then linking inputs to LCI datasets that characterize the production and delivery of those inputs. Inputs and field operations modeled in this study were based on University of California Cooperative Extension Cost of Production Studies for rice in the Sacramento Valley (Greer et al., 2012; Mutters et al., 2007). The cost studies are based on an annual cycle of activities, assuming a rice-only rotation, the use of high yielding medium grain varieties, and reflect farming practices considered typical for conventional rice producers in the Sacramento Valley area in California. The following sections describe the rice production systems, methods for modeling life cycle stages, how GHG emissions figures were derived, the methods for modeling co-products from rice production, and GWP calculations.

2.1. Field production practices

Rice production in California typically occurs on heavy clay soil using direct water-seeding practices. Land preparation begins in early April and involves initial plowing, followed by three passes with a disk, land leveling and rolling. Most N fertilizer is injected as aqua-ammonia before rolling the field. Other fertilizers (ammonium phosphate and zinc sulfate) are also applied as a "starter" blend to the soil surface by airplane either just before or after rolling (Table 1). Following these operations the fields are flooded and then seed is planted by airplane. After planting the fields are typically kept continuously flooded until about three weeks before harvest (typically in mid-September through early October) when fields are drained. During the first four to six weeks herbicides and pesticides are applied (mostly all by airplane) to ensure good crop establishment (see Section 2.4 for further information). At flowering, some farmers apply a fungicide to control blast and other diseases. Following harvest, the rice straw in the field is chopped and incorporated into the soil. The field is then flooded to facilitate straw decomposition (Linquist et al., 2006). During the winter, flooded rice fields also serve as important habitat for migrating water fowl (Central Valley Joint Venture, 2006). In late January through February, rice fields are drained. Irrigation systems in this region are gravity-fed from the Sierra Nevada mountain range, and thus we assumed no energy used in pumping. Farm equipment specifications are provided in Table 2.

The pesticides commonly used in CA rice systems to control weeds, insects and diseases are detailed in Mutters et al. (2007). The scarcity of life cycle inventory datasets for pesticides required us to select several surrogate pesticides for which life cycle inventory datasets are available. We replaced the four herbicides noted in Mutters et al. (2007) with three somewhat older products that target the same weed spectrum, based on information in the Pesticide Action Network (PAN) database (Kegley et al., 2009. Whenever possible, we chose products that are (or used to be) used in California rice systems with active ingredients from the same chemical family or that used a similar mode of action as the product being

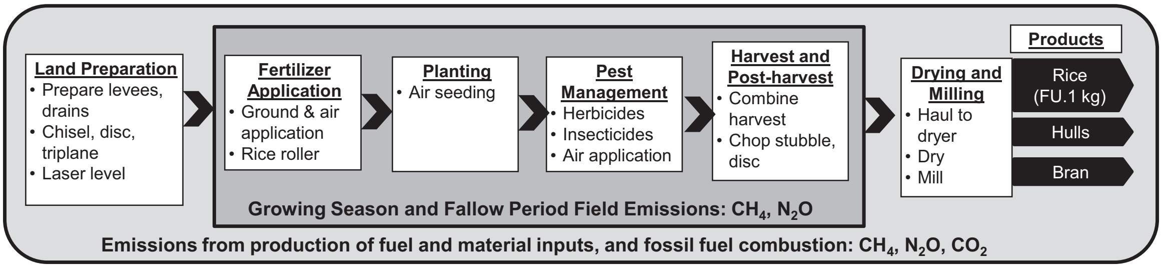

Fig. 1. Life cycle stages, sources of GHG emissions, functional unit (FU) and products for California rice production.

Table 2
Farm ground equipment specifications and percentage of area worked[a].

Operation	Equipment specification[b]	Operation time ha^{-1} or distance $load^{-1}$	% Of area
Maintain drains	Utility tractor, 67.5 kW	0.25 h	100
Maintain levees	4WD tractor, 168.75 kW	0.12 h	100
Chisel plow (2×)	4WD tractor, 168.75 kW	0.41 h each	100
Disc stubble	4WD tractor, 168.75 kW	0.57 h	100
Finish disc (2×)	4WD tractor, 168.75 kW	0.69 h each	100
Triplane	4WD tractor, 168.75 kW	0.25 h	100
Triplane	4WD tractor, 150 kW	0.25 h	100
Laser leveling	4WD tractor, 225 kW	1.24 h	25
Inject aqua ammonia	4WD tractor, 150 kW	0.72 h	100
Roll with rice roller	4WD tractor, 150 kW	0.67 h	100
Combine harvest	Combine, 187.5 kW	0.96 h	100
Bankout wagon	Self-propelled wagon, 187.5 kW	0.52 h	100
Mow levees	Utility tractor, 67.5 kW	0.05 h	41
Chop rice stubble	Utility tractor, 71.3 kW	0.67 h	100
Transport to dryer/mill	Truck, 22 Mt load	48 km	na
Transport agrochem. to farm	Truck, 1.8 Mt load	64 km	na
Transport fertilizers to farm	Truck, 22 Mt load	64 km	na

[a] *Sources:* Mutters et al. (2007); Greer et al. (2012).
[b] Fuel for all equipment is diesel.

substituted (Table 3). We also included soybean oil and a kerosene-based adjuvant as surfactants. LCI data for the fungicide Quadris, listed in the cost study, could not be found. Since it was only applied on a relatively small percentage of area and there is no alternative fungicide product registered for use in rice for the targeted diseases, we did not account for any fungicide application. Our final list of products was intended to represent a feasible list of active ingredients used in California rice and a reasonably representative number of aerial applications per season.

2.2. Post-harvest and milling of grain

Rice is harvested with a combine in September/October, when rice grains are on average 20% moisture, and then transported in bulk grain trailers to a drying facility, where it is dried to 14% moisture content using natural gas. Rice yields are reported after drying to 14% moisture and medium grain rice averages 9.3 Mt ha^{-1} (Greer et al., 2012). After drying, 90% of the paddy rice is milled to white rice and 10% to brown rice. We assumed the standard California electricity mix as the energy source for the milling equipment (Wang, 2009).

The rice production system generates co-products of hulls (20% of paddy rice weight) and rice bran (10% of paddy rice weight) along with finished rice. Because this study assesses LCGHG emissions for rice alone, a process is required to calculate the GHG emissions that are attributable only to rice, referred to as co-product allocation (see Section 2.5).

2.3. Field GHG emissions

Estimates for annual CH_4 and N_2O emissions from rice fields were derived from eight California field studies conducted from 1994 to 2011 in six different sites representing a range of typical rice production systems across the California rice producing region (Table 4). Each of these studies met the following criteria: (1) CH_4 fluxes were measured during the growing season under field conditions; (2) N fertilizer application rates were within the range of 100 to 165 kg N ha^{-1}, which is considered suitable for optimal yields; (3) the crop was water-seeded and was continually flooded throughout the growing season; and (4) rice straw was incorporated into the

Table 3
Pesticides and surfactants modeled in this study along with application rates and percentage of area applied[a].

Pesticide (active ingredient)	Pest controlled	kg ha^{-1} of product (kg ha^{-1} of active ingredient)	% Of total area
Bolero 8EC® (thiobencarb)	Weeds	4.4 (3.7)	100
Londax® (bensulfuron-methyl)	Weeds	0.078 (0.047)	100
2,4-D (Dimethylamine salt of 2, 4-D-dichlorophenoxyacetic acid)	Weeds	1.2 (0.58)	100
Warrior® (pyrethroid compound)	Weevil and armyworm	0.062 total (0.0069)	15 (1st applic.); 10 (2nd applic.)
Copper sulfate	Shrimp and algae	6.7	60
Crop oil (soybean oil)	Surfactant	8.6	100
Adjuvant (kerosene-based)	Surfactant	0.25	100

[a] *Sources:* Greer et al. (2012), Kegley et al. (2009) and Mutters et al. (2007).

Table 4
California studies providing methane and nitrous oxide field emission measurements. In all studies, GHG flux data were measured using chambers..

No.	Study	Location	Year	Growing season N rate ($kg\ ha^{-1}$)	Growing season CH_4 ($kg\ ha^{-1}$)	Winter season CH_4 ($kg\ ha^{-1}$)	Growing season N_2O ($kg\ ha^{-1}$)	Winter season N_2O ($kg\ ha^{-1}$)
1	Fitzgerald et al. (2000)	Maxwell1	1994–1995	135	67	64	na	na
		Maxwell1	1995–1996	135	106	168	na	na
2	Bossio et al. (1999)	Maxwell1	1997	150	106	na	na	na
3	Redeker et al. (2000)	Maxwell1	1998	150	61	na	na	na
		Maxwell1	1999	150	270	na	na	na
4	McMillan et al. (2007)	Maxwell2	2002	165	219	33	na	na
5	Pittelkow et al. (2014b)	Richvale1	2008	165	447	na	na	na
6	Pittelkow et al. (2013)	Arbuckle1	2010	140	107	45	0.35	0.47
		Arbuckle1	2011	140	208	59	0.27	0.37
7	Adviento-Borbe et al. (2013)	Robbins1	2011	100	na	0.07	na	0.37
8	Simmonds et al. (unpublished)	Richvale2	2011	130	92	na	0.09	na
Sample mean for seasonal emissions[a]					208	50	0.20	0.40
70% Confidence interval for mean seasonal emissions[b]					208 ± 75	50 ± 30	0.20 ± 0.70	0.40 ± 0.15
90% Confidence interval for mean seasonal emissions[b]					208 ± 135	50 ± 57	0.20 ± 0.22	0.40 ± 0.05
Annual mean					258		0.60	
IPCC Tier 1 (annual)					800		1.4	

[a] The sample mean first averages all values from a single location to reduce the bias of multiple data from a single site.
[b] Calculation of confidence intervals for the true mean emissions value assumes emissions samples are normally distributed and represent a random sample.

soil after harvest. In addition, growing and winter season N_2O fluxes were determined at three locations and winter fallow CH_4 emissions at four locations. All studies used the static vented chamber method (Hutchinson and Livingston, 1993) to measure GHG fluxes. Some studies were conducted at the same or in very close proximity to one another. To avoid any one location from over-influencing the results, we averaged all results from one location together first, and one average result from each location was then used to calculate our final average emissions over all locations, a method used in a previous study of rice field emissions in this region (Pittelkow et al., 2014a).

We did not account for any CO_2 emissions from the soil. Although soil CO_2 fluxes represent a source of short-term GHG emissions, in the long-run they are offset by high rates of net primary productivity and atmospheric CO_2 fixation by crop plants, and on a global scale are estimated to contribute less than 1% to the GWP of agriculture (Smith et al., 2007). In addition, we assumed that net carbon storage in the soil has stabilized in fields that have been in continuous flooded rice cultivation over the long-term.

Field emissions based on the Intergovernmental Panel on Climate Change (IPCC) Tier 1 methodology (IPCC, 2006) were also estimated, based on a total cultivation period of 150 days (from planting in April through harvest in September/October), assuming continuous flooding during the growing season, straw incorporation after harvest (>30 days before cultivation) equivalent to 9.3 Mt ha^{-1} (assuming a harvest index of 0.5) and flooding for greater than 30 days prior to the growing season.

2.4. Emissions from production of energy and materials

In order to model GHG emissions from the manufacture of fertilizers and pesticides, as well as rice seed production, we used data from Ecoinvent v.2.0, accessed through Simapro v.7 software (Pré Consultants, 2007; Ecoinvent Centre, 2008). The emissions are based on active ingredients only, which in some cases constitute only half of the pesticide product by weight. The identity of the inert ingredients, however, is considered proprietary information and is not publicly available. Detailed input quantities and LCI dataset names are provided in the Supplementary material (Section S1).

Transportation of agrochemicals from the retailer to the field site was included in the model. Because little to no information is available on the movements of agrochemicals, we assumed a distance of 64 km. Because of uncertainty in these values, we have included a sensitivity analysis on them. In baseline calculations, distances for trucking of the harvested rice to the drier are estimated at 48 km (personal communication with cost study authors). However, a study conducted in 1974 of energy use in California's agricultural sector estimated roundtrip distances of 167 km from field to processing site for rice, so this value was included in a sensitivity analysis of transport distances (Cervinka et al., 1974; Mutters et al., 2007). Due to lack of information, transport of co-products was not included in the baseline, but was included in sensitivity analysis. For transportation of agrochemicals from the manufacturer to retailer, we used emissions estimates as characterized in Ecoinvent 2.0 (Ecoinvent Centre, 2008). Many Ecoinvent datasets are based on European conditions, including distances from agrochemical manufacturer to retailer, which may be shorter than in the U.S. However, Kendall et al. (2012) mapped average U.S. freight rail and truck distances and calculated emissions based on these distances. We compared the results using these calculations with those using Ecoinvent datasets and found the differences to be negligible.

Equipment use emissions were modeled using the State of California's *OFFROAD* model based on activity data for field preparation, fertilizer applications, harvest, and straw incorporation, as detailed in the rice production cost study (California Air Resources Board, 2007; Mutters et al., 2007). *OFFROAD* provides data on criteria pollutant and CO_2 emissions, as well as fuel consumption per hour (California Air Resources Board, 2007). Upstream burdens for U.S. fuel production were also accounted for using data from Franklin Associates in the Simapro Software tool (Franklin Associates, 1998).

Most pesticides and some fertilizers are applied by airplane. Interviews with air applicators (Bob's Flying Service, personal communication 2008; Tolle's Flying Service, personal communication 2008) provided information on airplane load capacity, fuel use rates at full and empty capacity, and typical distances between base air strips and crop fields. Average fuel use required per ha for seeding, pesticide application, and fertilizer application were calculated based on the data collected during interviews and the density of the materials applied, application rates, and the percent of area requiring application (Supplementary material, Table S2).

CO_2 emissions were calculated directly from fuel use and non-CO_2 emissions were determined based on the airplane's engine power rating (562.5 kW). All of the flying services reported using Jet-A fuel for their planes. Because a Jet-A LCI dataset was not

Table 5
Co-product credits for milled rice co-products[a]. Cattle feed credits are based on a hay mix of 33% alfalfa and 67% ryewheat grass.

Co-product and use	Displaced pollutant		
	CO_2 (kg)	N_2O (kg)	CH_4 (kg)
Rice bran for cattle feed			
Per kg rice bran	0.241	5.54×10^{-6}	6.33×10^{-4}
Per ha rice bran	134	0.451	0.183
Hulls in electricity generation			
Per kg hulls	0.239	6.16×10^{-6}	7.04×10^{-4}
Per ha rice	444	0.0115	530
Hulls used in horticulture			
Per kg hulls	0.257	3.82×10^{-6}	1.71×10^{-4}
Per ha rice	477	7.11×10^{-3}	0.319

[a] Calculations based on information in Evans and Gachukia (2004), Forster et al. (1993) and Public Utilities Commission of the State of California (2008).

available, we used Ecoinvent's kerosene LCI as a surrogate (Ecoinvent Centre, 2008).

2.5. Co-product treatment

Rice bran and rice hulls resulting from the milling process have a number of uses in California. Rice bran is fed directly to cattle in an unstabilized form. Rice hulls have a number of uses, as bedding material for chickens and livestock, in horticulture where they replace vermiculite as a soil amendment in greenhouse applications and potting mixes, and for electricity generation in a rice hull power plant in Williams, CA. In this study we only considered rice hull use in horticulture (10% of hulls) and electricity generation (90% of hulls). The use of hulls as livestock bedding, which is a very low-value use, was not included in the analysis. Hulls used in this application have very low economic value, competing with waste materials from other industries. These factors make it difficult to assign an economic value to hulls used in this way or to determine avoided environmental impacts from production of alternative bedding materials.

The substitution, or displacement, method is used in this study to conduct co-product allocation. The substitution method examines products used in the same application as the co-products from a system and assigns a credit to the production system for generating co-products that avoid the production of these substitutable products. Rice bran was assumed to displace corn and a portion of the hay in cattle diets (Forster et al., 1993); hulls were assumed to displace the average fuel mix for an equivalent amount of electricity generated in California; and hulls were also modeled to displace expanded vermiculite as a soil amendment on a one-to-one basis. Further details are provided in the Supplementary material, Section S2, and the resulting co-product credits are reported in Table 5.

For comparison to the displacement approach, we also conducted an economic allocation, based on spot prices reported for rice, hulls, and bran at the mill (USDA, 2014). A detailed description of the data and results from this allocation approach is provided in the Supplementary material, Section S2.

2.6. GWP calculations

CO_2e emissions were calculated using GWPs from the IPCC Fourth Assessment Report's 100- and 20-year potentials (IPCC, 2007). By convention, most GHG inventories and assessments use 100-year GWPs (GWP_{100}), but the time horizon assumed for GWPs is particularly influential in assessing rice production because non-CO_2 gases, namely CH_4, dominate the inventory. CH_4 is particularly sensitive to the GWP time horizon because it is a short-lived gas compared to CO_2 and N_2O (IPCC, 2007).

A GWP standardizes the climate impact of a gas, as characterized by the ratio of cumulative radiative forcing (CRF) of a GHG relative to CO_2 calculated over a specified time horizon. CH_4 is a relatively short-lived gas (about 12 years) compared to CO_2, thus the 20-year GWP (GWP_{20}) of CH_4 is 76, while its GWP_{100} is 25 (IPCC, 2007).

3. Results and discussion

3.1. Field emissions

Methane emissions averaged 208 kg ha^{-1} (61 to 447 kg ha^{-1}) in the growing season and 50 kg ha^{-1} (0.07 to 168 kg ha^{-1}) during the winter fallow (Table 4). Methane emissions accounted for 97% of the GWP of annual field GHG emissions (6778 kg CO_2e vs. 177 kg CO_2 eq for N_2O). This is expected in systems that are flooded during both the growing and winter seasons. Under the anaerobic conditions of flooded fields, CH_4 is produced in the last step of organic matter degradation (Wassmann et al., 1998). The average amount of CH_4 emissions reported here is higher, but generally in line with, CH_4 emissions from rice systems globally of 133 kg ha^{-1} as reported by Linquist et al. (2012a). There is high variability in CH_4 emissions between studies; however such variability, even at the same site, is not unusual even under similar management (Pittelkow et al., 2013; Wassmann et al., 2000).

Nitrous oxide emissions averaged 200 g ha^{-1} (90 to 350 g ha^{-1}) during the growing season and 390 g ha^{-1}(370 to 470 g ha^{-1}) during the winter fallow. Under anaerobic conditions, N_2O emissions are negligible as most N_2O produced in these systems is further reduced and lost as N_2 (Firestone and Davidson, 1989). Thus N_2O emissions are typically low in flooded rice systems (Akiyama et al., 2005) compared to other cropping systems. N_2O emissions were observed in these studies after soils were drained for harvest until fields were flooded for the winter, and following the drainage at the end of the winter fallow. In these studies, emissions of either CH_4 or N_2O during the winter were typically low (with the exception of Fitzgerald et al., 2000), due to low temperatures, regardless of whether the field was flooded or not (Sass et al., 1991; Gödde and Conrad, 1999).

Table 4 shows 70% and 90% confidence intervals for mean emissions for each emissions category (growing season and winter emissions for CH_4 and N_2O, respectively). These are *t* statistic confidence intervals, which assume that emissions within each seasonal group are randomly sampled and normally distributed.

Using the IPCC Tier 1 calculations to estimate field GHG emissions resulted in annual field CH_4 emissions of 800 kg ha^{-1} and N_2O emissions of 1.4 kg ha^{-1}, which are substantially higher than the average emissions calculated from the eight field studies. In California, heavy clay soils and the fact that most of the added N is injected below the soil surface might be contributing to lower CH_4 emissions per area than is being accounted for in the global data used to construct the IPCC emissions factors (Huang et al., 2002; Jäckel et al., 2001; Linquist et al., 2012a).

3.2. Baseline life cycle emissions

Based on average yields of 9.3 Mt ha^{-1} (Greer et al., 2012), the baseline net LCGHG emissions are 1.47 kg CO_2e kg^{-1} of milled rice (Table 6). Field emissions are the largest contributor to life cycle GHG emissions, at 1.02 kg CO_2e kg^{-1} rice, or 69% of net life cycle CO_2e emissions, based on GWP_{100} (Fig. 2). Fertilizer production (including manufacture and distribution) is the second largest contributor, at 0.28 CO_2e kg^{-1} milled rice (19% of total net emissions). All on-farm ground equipment operations, including land preparation, aqua-ammonia injection, harvest, and straw management, together contribute 11% to the total (0.17 kg CO_2e kg^{-1} milled rice),

Table 6
GHG and CO_2e emissions for baseline rice production processes.

Operation	N_2O kg ha^{-1}	CH_4 kg ha^{-1}	CO_2 kg ha^{-1}	kg CO_2e kg^{-1} rice (GWP_{100})	kg CO_2e kg^{-1} rice (GWP_{20})
Land preparation	4.54×10^{-5}	0.0861	404	0.0624	0.0630
Fertilizer production[a]	0.0287	3.38	1760	0.284	0.311
Fertilizer application[b]	3.78×10^{-4}	0.0445	193	0.0298	0.0302
Seed production and seeding	0.117	0.0842	67.9	0.0161	0.0166
Pest management[c]	0.0239	0.0925	46.7	8.61×10^{-3}	9.31×10^{-3}
Annual field emissions	0.595	258	0.00	1.02	3.04
Harvest and straw management	5.72×10^{-5}	0.0969	508	0.0784	0.0792
Transportation[d]	3.34×10^{-6}	0.0102	37.3	5.77×10^{-3}	5.85×10^{-3}
Drying and milling	6.54×10^{-3}	1.40	495	0.0816	0.0925
Total	0.772	264	3510	1.59	3.65
Co-product credit	−0.462	−1.36	−573	−0.116	−0.124
Total net	0.310	262	2940		
Total net (kg CO_2e)	**92.4**	**6550**	**2940**	**1.47**	**3.52**

[a] Includes fertilizer manufacture and delivery to retailer.
[b] Includes ground equipment and airplane operations.
[c] Includes pesticide manufacture, delivery to retailer, and application by airplane.
[d] Includes transport of inputs from retailer to farm and transport of paddy rice to drying and milling.

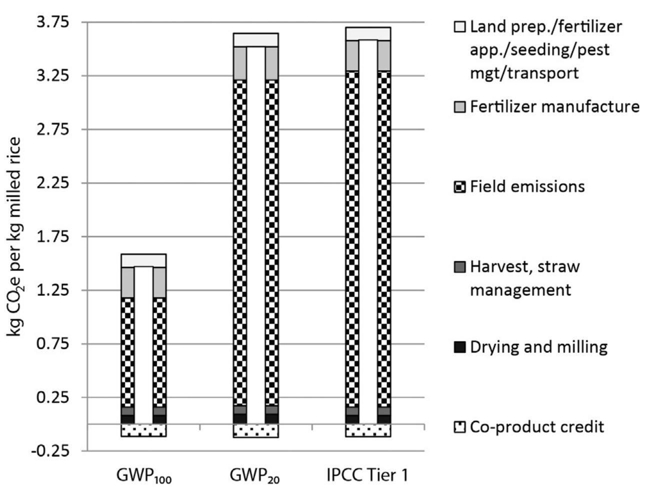

Fig. 2. Contribution by life cycle stage to total CO_2e emissions for milled rice production, including baseline results using 100-year and 20-year global warming potentials (GWP), and results using IPCC Tier 1 estimates for field CH_4 and N_2O emissions (using GWP_{100}).

with land preparation comprising 37% of that contribution, or 4% of total net life cycle emissions. While small, co-product credits are not negligible, accounting for 8% of net emissions on an absolute value basis. Using an economic allocation method instead of the displacement method resulted in slightly more emissions being allocated to rice, with co-product credits assigned 4 to 5% of net emissions, resulting in a change of net life cycle emissions for rice of 3 to 4% (details available in the Supplementary material, Section S2).

Pest management, drying and milling, and transportation were all small contributors to LCGHG emissions. In fact, manufacture and application of pesticides contribute only half of 1% of total emissions. However, pesticide active ingredients vary tremendously in their GHG emissions, due to different chemical formulations and varying application rates.

The herbicide ingredient thiobencarb accounts for over half of the total CO_2e for pest management, and crop oil accounts for another 18%, as both products are applied at higher rates than the other materials. Due to our substitution of older herbicide products, some of which do not require any adjuvants, our application rate of 8.6 kg ha^{-1} of crop oil is equivalent to only half of the rate noted by Mutters et al. (2007). Applying the full amount noted in the cost study would increase the total emissions from pest management by 15%. Thus, choosing pesticides that are effective at low application rates and that do not require adjuvants such as crop oil may lower the GHG emissions associated with pest management.

3.3. GWP time horizon and CO_2e calculations

Using GWP_{20} results in a figure for CO_2e emissions of 3.52 kg per kg of milled rice, 233% higher than the figure using GWP_{100} (Fig. 2). Field emissions also contribute a substantially larger portion of total emissions, 86%, when a GWP_{20} is used, compared to 69% when a GWP_{100} is used. Though 100-year GWPs are typically used to characterize non-CO_2 GHGs in policies and standards, the dramatic effect of considering shorter time horizons suggests that assessments of rice should include GWPs calculated at both 20 and 100 year time horizons.

3.4. Comparisons with other estimates of life cycle GHG emissions for rice production

Four recent studies as well as the Ecoinvent life cycle inventory database have characterized life cycle emissions of rice production in different regions (Table 7). Two of these, Hokazono and Hayashi (2012), in Japan, and Ecoinvent Centre (2008), reported results similar to or lower than our study, at 1.46 kg CO_2e kg^{-1} milled brown rice (for the conventional production system they model) and 0.47 kg CO_2e kg^{-1} rice at farmgate, respectively. Hokazono and Hayashi used IPCC Tier 1 methodology to estimate field CH_4 emissions, resulting in 22% lower CH_4 emissions on an aerial basis than our study. Their system is characterized by intermittent field aeration during the growing season and non-flooding before the season, two factors that reduced their emissions estimates from the IPCC default figures. They also used only half of the N fertilizer application (average 82 kg ha^{-1}) typically used in California, with IPCC Tier 2 methods to estimate N_2O emissions. However, their yields were only 46% of our baseline yields. The Ecoinvent LCA provides very little detail on the analyzed system, but indicates biotic methane emissions an order of magnitude lower, suggesting either a very different scope for emissions accounting (e.g. only growing season), an error in calculations, or a lack of accounting for field emissions entirely.

The other three studies, Blengini and Busto (2009) in Italy, Thanawong et al. (2014) in northeastern Thailand, and Wang et al. (2010) in central China, report from 47% to 544% higher life cycle emissions than our study, even after adjusting for system

Table 7
Comparison of this study's grain yields, N fertilizer inputs, and life cycle greenhouse gas emissions results with recent studies from other regions.

Study authors and region	Yield	N fertilizer	Total net LCGHG to farmgate	Total net LCGHG with milling	Other notes
	$Mt\ ha^{-1}$	$kg\ N\ ha^{-1}$	$kg\ CO_2e\ kg^{-1}$ rice		
This study (measured field emissions), California	9.3	107	1.01	1.55	Functional unit includes 90% white, 10% brown rice
This study (IPCC Tier 1 emissions), California	9.3	107	1.09	3.29	Functional unit includes 90% white, 10% brown rice
Hokazono and Hayashi (2012), Japan	4.3	82	na	1.46	Functional unit includes 100% brown rice
Ecoinvent Centre (2008), Unknown	7.5	?	0.47	na	Very low CH_4 emissions suggests omission of field emissions
Blengini and Busto (2009), Italy	6.1	128	na	2.52–2.66	Milling and packaging are combined: low figure is without this stage, high figure is including this stage
Thanawong et al. (2014), NE Thailand	2.2	?	2.97–5.55	na	No details provided on fertilizer analysis or quantity
Wang et al. (2010), central China	8.8	318	1.50	na	No details provided on field emissions

boundary differences. The study by Wang et al. (2010) is the only one with yields (8.8 $Mt\ ha^{-1}$) similar to those in this study, but used N fertilizer rates almost double those in California (318 $kg\ N\ ha^{-1}$), which may account for the higher life cycle emissions. However, details about field emissions are not provided and may also be an important source. The yields of the other two studies range from 24% (Thailand) to 66% (Italy) of this study's yields, and their field methane emissions range from approximately 30% to 80% higher on a per area basis. Both studies used IPCC Tier 2 guidelines for calculating emissions, utilizing published country-specific (Italy) or region-specific (northeastern Thailand) emissions factors that were based on field measurements. Field emissions accounted for 62% and 68% of total life cycle GWP for the Thai and Italian studies, respectively. This proportion is similar to the 69% in this study (not accounting for slightly different system boundaries). Together, these five studies illustrate the importance of field methane emissions in influencing net LCGHG emissions, as well as the role of high yields in scaling down the final results on a mass basis. With the highest yields and some of the lowest per area field emissions, the typical California production system modeled in this study compares favorably to other rice producing regions, except in cases such as the Japan study, in which periodic field drainage reduced emissions.

How field emissions are estimated can play a large role in shaping the net results. Using the IPCC Tier 1 calculations in our model instead of the field-measured average emission figures resulted in much higher total net GWP_{100} of 3.58 $kg\ CO_2e\ kg^{-1}$ milled rice, with field emissions accounting for 87% of the total. This figure for net LCGHG emissions is closer to those found in several of the studies above, although the percent contributed by field emissions is much higher than in those studies. This result indicates that the IPCC Tier 1 method greatly overestimates emissions for California conditions, in contrast to the northeastern Thailand context, where it underestimates emissions (Thanawong et al., 2014).

3.5. Fallow season emissions

Attribution of winter season field emissions to the rice system could be questioned, on the grounds that most of the rice growing area in northern California occupies areas that, due to impermeable soils, were historically seasonal wetlands during the winter rainy season, prior to the advent of industrial-scale commercial farming. Therefore, emissions of GHGs from flooded fields in the winter might arguably be considered as natural background emissions that are separate from human-induced emissions from rice farming (ignoring any major differences in vegetation cover). The winter fallow contributes 66% of annual field N_2O emissions, and 19% of annual CH_4 emissions, which together amounts to 14% of total net life cycle CO_2e emissions. Without these winter emissions, the total life cycle emissions for the system are reduced to 1.26 $kg\ CO_2e\ kg^{-1}$ milled rice, or 86% of the baseline total. The field emissions comprise 64% of this reduced total, having been reduced from 6631 $kg\ CO_2e\ ha^{-1}$ to 5259 $kg\ CO_2e\ ha^{-1}$. Therefore, how to characterize the "natural background" versus rice system emissions, and where to draw the boundaries, is an important consideration. In a review of studies of rice paddy and natural wetlands methane emissions in China, Chen et al. (2013) found that overall methane fluxes from natural wetlands tended to be lower, on a per area basis, than those from rice paddies, but they did not directly compare flooded fallow season emissions from rice paddies with natural wetland emissions during the same time of year.

3.6. Sensitivity analyses

Sensitivity analysis was conducted on the largest sources of emissions and on those parameters considered most uncertain or influential on study outcomes. Field emissions and N fertilizer use contributed the most to net emissions, and field emissions and transportation distances had the highest uncertainty. In addition, grain yields strongly influence the final results.

3.6.1. Field GHG emissions

The sensitivity analysis on field emissions used the 70% and 90% confidence intervals around the mean seasonal CH_4 and N_2O emissions (Table 4). The 70% confidence interval resulted in field emissions of ±43% of the mean annual CO_2e emissions, and LCGHG emissions of ±30% of the baseline results (LCGHG of 1.03–1.91 $kg\ CO_2e\ kg^{-1}$ rice). The 90% confidence interval led to a

±74% change in field emissions, and approximately a ±50% change in the baseline result (LCGHG of 0.72–2.22 kg CO_2e kg^{-1} rice). These results confirm that field emissions strongly influence the life cycle emissions of this rice production system. However, the overall LCGHG estimation was still relatively low compared to many of the other studies, and also when compared to using IPCC Tier 1 methodology.

3.6.2. Yield

We performed a sensitivity analysis on yield, increasing and decreasing the baseline yield of 9.3 Mt ha^{-1} by one standard deviation (0.42 Mt ha^{-1} post-drying weight, or about 4.5%), based on interannual variation in statewide yield data from 2001 to 2012 (CDFA, 2013). Using this variation, the net LCGHG emissions ranged from 1.40 to 1.54 kg CO_2e kg^{-1} milled rice (±4.8% around the baseline).

3.6.3. Transportation distances

We conducted a sensitivity analysis for transportation distances due to uncertainty in distances for both inputs and outputs for the rice production process. The sensitivity analysis varied agrochemical transport distances from 64 km to 96 km; transport distances for harvested material to processing sites from 48 to 167 km, and added co-products transport distances from 48 to 161 km. Due to lack of reliable data, we did not include any emissions for co-product transport in our baseline calculations. Adding emissions for the shortest co-product distance (64 km) resulted in an increase of 28% in total transportation-related emissions, while the longest distance (161 km) resulted in a further increase of 69%. However, the maximum increase in CO_2e emissions from transportation, 96%, occurred when varying transport of harvested rice from field to drier. However, even with this large increase, total transportation–related emissions only rose from 0.39% to 1.2% of total net life cycle emissions, and the total LCGHG emissions rose by less than 1%. Therefore, uncertainty in these distances is unlikely to substantially affect modeling outcomes.

3.6.4. Fertilizer rates

Because not all growers apply potash and zinc sulfate, we conducted sensitivity analyses for each of these inputs to examine the effects of no input. Eliminating each input one at a time resulted in reduction of 0.32% (in the case of potash) or 0.16% (in the case of zinc sulfate) of total life cycle emissions per kg of finished rice.

In contrast, varying N fertilizer application rates has a much larger impact on life cycle emissions. We adjusted our baseline model to include the fertilizer rates and field emissions data from a study conducted by Pittelkow et al. (2013). The two-year study included two representative fertilizer rate treatments – one of 140 kg N ha^{-1}, and the other of 200 kg N ha^{-1} – with both treatments relying exclusively on aqua ammonia for their N source. CH_4 and N_2O were measured during both the growing and winter seasons. All other inputs as well as rice yields were held constant as described for the baseline model. Pittelkow et al. (2013) observed no significant yield differences as a result of changes in fertilizer rates, with yield being stable between140 and 200 kg N ha^{-1}. Optimal N rates vary among fields for a variety of reasons but fertilizer N rates of 140–200 kg N ha^{-1} are typical for California (Linquist et al., 2009). Results indicate that, while field emissions of N_2O and CH_4 increased by 26% and 3%, respectively, emissions of N_2O, CH_4, and CO_2 from fertilizer manufacturing increased by 39%, 40%, and 42%, respectively, at the higher fertilization rate. LCGHG emissions per kg of milled rice increased by approximately 10% when using the higher N application rate compared to the lower rate. While meta-analysis of research on fertilization practices and GHG emissions in rice production has shown that, within the agronomically optimal range of 100 to 200 kg N ha^{-1}, there is limited effect of fertilizer N rate on CH_4 emissions (Linquist et al., 2012a; Pittelkow et al., 2014b), analysis here shows that the effects on upstream emissions can be substantial and over fertilization of N should be avoided.

3.7. Implications for life cycle emissions reductions

Although California rice production systems use more industrial inputs (including fertilizer and heavy field equipment) on an areal basis than many other systems, they also achieve much higher yields, thus creating lower impacts per unit of output. This analysis suggests that priority should be placed on continued improvements in input-use efficiency in all systems. It also demonstrates the significance of scaling environmental impacts by crop yield.

In terms of reducing emissions occurring throughout the system, our results indicate that field emissions (especially of CH_4), fertilizer inputs, and use of equipment for land preparation, harvest and transport constitute the major emissions hotspots. Attempts to reduce life cycle emissions in California rice production, therefore, should target these areas. Periodic drainage of rice fields during the growing season has been investigated as a method to substantially reduce CH_4 emissions (Chen et al., 2013; Yan et al., 2009). Early-season drainage is starting to be more widely adopted by rice growers in California to facilitate application of new foliar-active herbicides, but this practice may also increase N_2O emissions by leading to buildup of nitrate-N in the soil (Linquist et al., 2012b). However, using IPCC emissions factors on large regional scales, Yan et al. (2009) estimated that the reduction in GWP from CH_4 reductions would outweigh the smaller increases resulting from higher N_2O emissions. They also note that incorporating straw during the off-season rather than immediately prior to the next cultivation cycle could also reduce CH_4 emissions, but this is already a common practice in California.

Maximizing fertilizer use efficiency and applying only the amount of N required by the crop for that season is also an important avenue to lowering life cycle emissions. Our results demonstrate that reduction of fertilizer rates not only reduces field emissions (especially N_2O emissions, which could increase with more adoption of in-season drainage), but also reduces upstream emissions to an even larger extent, due to the substantial CO_2e emissions associated with synthetic N fixation and fertilizer formulation. Although growers already have incentives to optimize fertilizer use efficiency to save on input expenses, research suggests that fertilizer application rates typically vary widely among rice growers (Linquist et al., 2009), demonstrating a greater need for research and extension efforts to clarify N needs for different soil types, rice varieties, and flooding regimes. In addition, discussions with rice growers indicate uncertainty about the availability of N from rice straw. Linquist et al. (2006) indicate that winter straw incorporation can potentially increase early-season N availability for the next crop by 19 to 25 kg N ha^{-1} in some soil types, which could lead to a 15% savings in N fertilizer applications. A reduction of this magnitude in aqua-ammonia application in our modeled system (assuming no change in yields) would result in a net total LCGHG reduction of 2.8%. Other fertility management practices, such as deep placement of N (as is done with pre-flooding injection of aqua ammonia by California growers) and use of enhanced-efficiency fertilizers, have also been shown to reduce field emissions (Linquist et al., 2012a), although the cost implications of these products for growers need to be examined.

Finally, increasing the fuel efficiency of farm machinery and reducing the number of tillage passes are two factors that could provide additional opportunities for small but significant GHG savings, given that use of farm equipment alone comprises 11% of the baseline result, with land preparation alone comprising 4%. Linquist et al. (2008) have explored the potential for adoption of a minimum tillage regime in California rice production, but they also suggest that doing so may require higher N inputs, due to more denitrification losses. These tradeoffs need further research and analysis of net impacts on LCGHG emissions.

4. Conclusions

Due to inconsistencies in system boundaries and assumptions, direct and conclusive comparison across life cycle GHG emissions studies for rice production is difficult. However, based on the assumptions and system boundaries used in this study, California rice production seems to be less emissions-intensive per kg of rice, than most other rice producing regions, likely due to high grain yields combined with relatively low field CH_4 emissions. These results suggest that intensive production systems that produce high yields do not necessarily always result in more environmental harm than low-input systems, especially if inputs are optimized in relation to achievable yields, and when situated in suitable production regions whose physical characteristics, such as soil type and climate, help to maximize production efficiencies.

The dominance of field emissions indicates that significant reductions in GHG emissions from rice production must target this source, particularly CH_4. Given the large variability in measured values of field emissions, as well as the substantial differences between these values and estimates derived using the IPCC Tier 1 approach, future research should focus on improved characterization of these emissions for different soil conditions and with different crop management practices. Research is also needed to integrate these findings into management practices that maintain or increase yields and/or net profit for growers while reducing field emissions. Given that many crop management practices, including straw management, water management, and synthetic N fertilizer use, incur GHG emissions in other parts of the system in addition to affecting field emissions, understanding the full life cycle GHG implications of changes in management practices and in materials used can assist growers, researchers, food buyers, and policy makers in choosing the most effective practices for reducing GHGs from rice production.

Finally, this study demonstrates that the use of 100-year GWPs to report CO_2e emissions strongly influences the outcome of rice's 'CO_2e footprint'. This indicates that footprints and other summary indicators for climate change effects should choose the GWP time horizon extremely carefully for rice and other products with a high proportion of CH_4. This choice of time horizon may be particularly important when policies or decisions are focused on achieving near-term climate change mitigation targets, or when the irreversibility of climate change processes and effects is considered.

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Appendix A. Supplementary data

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