Title

Effect of a year-long fallow on carbon-nitrogen cycling in California continuous rice systems and perspectives on net system greenhouse gas emissions accounting

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Effect of a year-long fallow on carbon-nitrogen cycling in California continuous rice systems and perspectives on net system greenhouse gas emissions accounting

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From then till now, I never stopped dreaming of feeding the world. Will studying soil phenols and methane do that? I know not – but opportunities come and I took it. Part of the practice is to keep going at it until it all makes sense.

And now that I appear to be a little more learned, I hope that the future you will not forget why you started. I aspire that you will be truly open and accepting of new ideas, that you will keep learning and thinking, that you remember science does not exist in a silo, but should be in conversation with the world around you, and most importantly, that you will hold compassion and empathy close to your heart. Take with you loving kindness, and forget not, the humanity that we share, and the promise you once made of feeding the world.

Abstract

Rice (*Oryza sativa* L.) is integral in meeting global food and nutritional demand. However, its production is becoming increasingly challenging due to climate change and its associated impacts. In this dissertation, we explore the carbon (C) and nitrogen (N) cycling dynamics in rice systems in the context of climate change. Chapter one examined opportunities for greenhouse gas (GHG) emissions mitigation and accounting in Southeast Asia (SEA) through a systematic review. Chapters two to four examined the effect of a year-long fallow on soil N availability, GHG emissions, and agronomic productivity when introduced to continuous rice systems in California.

In chapter one, the review synthesized findings across four main components of net system emissions: (1) field GHG emissions, (2) energy inputs, (3) residue utilization beyond the field, and (4) SOC change. Integrating all four components there were two main takeaways. First, the components of field GHG emissions and SOC change were the biggest opportunities for reducing net system emissions and need to be considered for effective climate change mitigation. Second, the reduction of C inputs through residue removal and increased soil aeration through multiple drainage will lower methane (CH_4) emissions but may also decrease SOC stocks over time. Hence, we argue that future research needs to consider cross-component effects to optimize net system emissions, specifically the "stacking" of best management practices for mitigation related to field GHG emissions or SOC change in long-term experiments.

In chapters two to four, we examined the effect of fallow interjection on continuous rice systems in California. Due to winter droughts leading to water restrictions or spring rains leading

to prevented planting, growers have been forced to fallow their lands. As such, the norm of continuous rice cropping is challenged and the year-long fallow will have important effects on soil N fertility, GHG emissions, and agronomic productivity. There were two main treatments – continuous rice (CR) and fallow rice (FR; rice following a year-long fallow). In chapter two, we evaluated crop uptake of soil N (N $uptake_{soil}$) and fertilizer N (N $uptake_{fertilizer}$) using ^{15}N-enriched ammonium sulfate. Examining the sources of crop N uptake, N $uptake_{soil}$ in the FR treatment was 16.7 kg N ha^{-1} higher than the CR treatment at maturity but N $uptake_{fertilizer}$ was similar. These results indicate that N $uptake_{soil}$ was primarily responsible for lower N uptake in CR. Soil phenols, which have been documented to accumulate in continuously flooded rice systems and stabilize soil N, were greater in CR than FR in both the research station study and the regional survey study. Together, higher phenol levels and lower N $uptake_{soil}$ in CR provide mechanistic evidence that the introduction of a season-long fallow to continuous rice systems enhances soil N availability by reducing organic substrate recalcitrance.

In chapter three, we quantified GHG emissions (CH_4 and N_2O) of the systems, including the year-long fallow (F). In the summer, FR had a 45-53% reduction in cumulative CH_4 emissions compared to FR in two of three years. F had no CH_4 emissions in the summer. Summer N_2O emissions were low for all three treatments. Summer global warming potential (GWP) accounted for more than 96% of annual GWP in CR (13,937 kg CO_2 eq ha^{-1}) and FR (9,236 kg CO_2 eq ha^{-1}). For F, the winter season accounted for 94% of the annual GWP (413 kg CO_2 eq ha^{-1}) due to N_2O emissions. Additionally, we showed that particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) levels in CR and FR were similar in the year with no treatment effect on CH_4 emissions.

In the final chapter, we evaluated the agronomic productivity of the two systems. Maximum observed yields did not differ between CR and FR, averaging 14.0 Mg ha^{-1} in 2021, 12.6 Mg ha^{-1} in 2022, and 9.6 Mg ha^{-1} in 2023. Based on quadratic regressions of yield response to N, the agronomic optimum N rate (AONR) was higher for CR in all years. Where no fertilizer N was applied, FR yielded higher than CR, averaging a difference of 2.9 Mg ha^{-1}. The yield differences at 0 kg N ha^{-1} can be attributed to soil N availability, where FR averaged 31.6 kg N ha^{-1} more soil N uptake than CR at maturity. Apparent fertilizer nitrogen recovery efficiency (FNRE) did not differ between treatments and averaged 59.8%. Stem rot, caused by *Sclerotium oryza*e, was more severe in CR than in FR, having averaged severity indexes of 3.7 and 3.1 respectively. Based on differences in soil N uptake and FNRE, the N rate can be reduced by roughly 50 kg N ha^{-1} for fields following a fallow compared to continuously cropped fields, allowing growers to maintain yields with lower inputs.

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Chapter One

Title:

Opportunities for mitigating net system greenhouse gas emissions in Southeast Asian rice production: A systematic review

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Highlights

- Net system emissions include field CH_4/N_2O emissions, energy inputs, residue utilization, and SOC change
- Integrated review highlights cross-component effects of management including tradeoffs
- Field GHG emissions and SOC change are the largest opportunities for mitigation
- Reduced C inputs and drainage decrease CH_4 emissions, but may negatively affect SOC
- "Stacking" of best management practices is a key knowledge gap that warrants research

Abstract

Southeast Asia (SEA) is a key producer and exporter of rice, accounting for around 28% of rice produced globally. To effectively mitigate greenhouse gas (GHG) emissions in SEA rice systems, field methane (CH_4) and nitrous oxide (N_2O) emissions have been intensively studied. However, an integrated assessment of system-level GHG emissions which includes other C balance components, such as soil organic carbon (SOC) or energy use, that can positively or negatively influence the net capacity for climate change mitigation is lacking. We conducted a systematic review of published research in SEA rice systems to synthesize findings across four main components of net system emissions: (1) field GHG emissions, (2) energy inputs, (3) residue utilization beyond the field, and (4) SOC change. The objectives were to highlight effective mitigation opportunities and explore cross-component effects to identify tradeoffs and key knowledge gaps. Field GHG emissions were the largest contributor to net system emissions in agreement with existing scientific consensus, with results showing that practices such as floodwater drainage and residue removal are sound options for CH_4 mitigation. On the other hand, increasing SOC potentially provides a large GHG mitigation opportunity, with long-term continuous rice cropping and practices such as residue incorporation and biochar application promoting SOC increase. A reduction in energy inputs was mainly achieved by optimizing agrochemical use, especially N fertilizers. For residue utilization beyond the field, GHG emission mitigation mainly came from preventing open field burning through residue removal. Removed residue can subsequently be used for producing energy that offsets GHG emissions associated with conventional fuel sources (e.g. fossil fuel-based electricity generation) or substituting material used in other production systems. Integrating all four components of net system

emissions into one analysis underscores the following two main takeaways. First, the components of field GHG emissions and SOC change are the biggest opportunities for reducing net system emissions and need to be considered for effective climate change mitigation. Second, the reduction of C inputs through residue removal and increased soil aeration through multiple drainage will lower CH_4 emissions but may also potentially decrease SOC stocks over time. Hence, we argue that future research needs to consider cross-component effects to optimize net system emissions, specifically the “stacking” of best management practices for mitigation related to field GHG emissions or SOC change in long-term experiments.

Keywords

Greenhouse gas, soil organic carbon, energy input, residue and water management, climate smart agriculture

1. Introduction

Rice is an important crop in Southeast Asia (SEA), serving both as a key source of caloric intake and economic livelihood (Redfern et al., 2012). The world produced a total of 782 million tonnes of rice in 2018 - of which 28% (220 million tonnes) was produced in SEA (FAOSTAT, 2020). In particular, Indonesia, Vietnam, Thailand, Myanmar, and the Philippines represent 5 out of 10 of the world's largest producing countries, and account for 92% and 91% of area harvested and production respectively within SEA (FAOSTAT, 2020). Tropical rice systems are facing the challenge of not only increasing crop productivity but also improving resource-use efficiencies related to water, energy, and agrochemical inputs (Yuan et al., 2021). Moreover, because rice cropping systems are the dominant form of agricultural land use in SEA, it is critical to address growing environmental concerns related to greenhouse gas (GHG) emissions and carbon (C) footprint, which are often associated with high water and energy consumption, and fertilizer and pesticide pollution (Wassmann, 2019).

Compared to other staple food crops, flooded rice systems play a more prominent role in global agricultural GHG emissions (Smith et al., 2008). It has been estimated that rice accounts for roughly half of total global crop production emissions in terms of carbon dioxide (CO_2) equivalents per kilocalorie produced (Carlson et al., 2017). Two recent developments in international policy and trade make rice systems in SEA an especially key player in climate change mitigation. First, several countries including Vietnam and Indonesia have committed to the Paris Agreement, an international treaty on climate change requiring them to take action on reducing GHG emissions to prevent global warming (Tran et al., 2019). National GHG inventory data for SEA indicates that rice systems contribute on average 20% of total emissions at the country level

(Wassmann, 2019), highlighting the importance of mitigation opportunities in agriculture from a policy and government perspective. Second, rapid changes are occurring in the commercial sector to improve the sustainability of global rice supply chains. Since SEA is a leading rice exporter, efforts to track and mitigate net system GHG emissions are increasingly implemented at the farm level (Devkota et al., 2019). An improved understanding of the different factors contributing to net system emissions would help inform the development of public and private sector mitigation programs.

Methane (CH_4) and nitrous oxide (N_2O) are the primary sources of GHG emissions in rice systems - especially CH_4 caused by high C inputs (rice roots and residues) decomposing under anaerobic conditions in flooded soils (Le Mer & Roger, 2001). A large body of research has demonstrated that GHG reduction can be achieved through reducing C inputs or water management strategies that reduce the period of flooding during the growing season, often through field drainage events (Feng et al., 2013; Haque et al., 2020; Jiang et al., 2019; Setyanto et al., 2018). Recently, Yagi et al. (2020) showed in a meta-analysis of the SEA region that CH_4 emissions can be significantly reduced (35%) through single or multiple drainage events such as alternate wet-dry (AWD) irrigation practices. Other strategies such as rice straw removal, soil drying during the fallow period, and application of biochar were also documented as promising strategies to mitigate CH_4 emissions - although more research is required for some of the options examined (e.g. long-term effects of biochar application). While Yagi et al. (2020) consolidated region-specific evidence on mitigating field GHG emissions, additional studies have been published since, and results were not discussed in relationship to other components of C cycling that can impact net system emissions.

A singular focus on reducing field GHG emissions is an incomplete picture of climate change mitigation in rice systems as it fails to consider other C sources or sinks such as energy consumption and changes in soil organic carbon (SOC) (Zhang et al., 2017; Liu et al., 2014; Shang et al., 2021; Silalertruksa & Gheewala, 2013). Analysis of direct and indirect energy use in crop production is required to account for the embodied energy in external inputs such as nitrogen (N) fertilizers and fuel use by machinery (Lal, 2004). These inputs can be converted into CO_2 equivalents using life cycle analysis (LCA) methodology and compared to other sources of emissions (Sieverding et al., 2020). Nguyen et al. (2019) in a study in the Philippines reported that field GHG emissions represented the highest proportion of total emissions (63–84%) followed by mechanized operations, fertilizer, and in-field burning of rice residue, accounting for 9-15%, 6-11%, and 11% of total emissions, respectively. By understanding the energy inputs of rice production and key factors influencing efficiency, management can be fine-tuned for reduced emissions by manipulating synthetic fertilizers and energy usage (Zhang et al., 2017).

Residue management also influences net system emissions. Rice has a harvest index of roughly 50% (Yang & Zhang, 2010), creating large amounts of C-rich crop residues that serve as substrate for CH_4 production. Concerning net system emissions, there are three main options for residue management: removal from the field, in-field burning, or incorporation into the soil. Residue removal and utilization beyond the field has several potential benefits including the production of fuel or energy (Silalertruksa et al., 2013; Silalertruksa & Gheewala, 2013), or substituting material usage in other production systems such as bedding in mushroom cultivation (Nguyen et al., 2019). In addition, residue removal can help mitigate net system emissions by reducing field GHG emissions (e.g. preventing increased CH_4 emissions from higher C inputs due to residue incorporation) (Liu et al., 2014; Romasanta et al., 2017), as well as avoiding GHG emissions associated with in-field burning of residues (Wassmann, 2019). However, residue incorporation also provides an important source of C to maintain soil fertility and SOC stocks in the long term (Chivenge et al., 2020), thus residue removal may have tradeoffs for SOC. Therefore, scientific frameworks for net system emissions must account for the benefits and costs of residue management across these different components.

Finally, rice soils hold the potential to mitigate climate change as they are a large pool of C stock, and associated SOC increases have huge potential to reduce net GHG emissions (Amelung et al., 2020; Liu et al., 2021). Whether SOC increases or decreases in paddy soils in response to management practices such as straw removal or intermittent irrigation would strongly influence net system emissions. To determine the net GHG balance of different water and C management strategies, several studies have developed new insights by integrating CH_4 and N_2O emissions with corresponding SOC change (Liu et al., 2014; Shang et al., 2021). For example, the global rice community considers the practice of AWD to be effective for reducing field GHG emissions, but research has questioned whether a higher frequency of non-flooded soil conditions might decrease SOC to a greater extent, leading to an overall increase in net system emissions (Livsey et al., 2019). In contrast, evidence from other cereal systems suggests that positive SOC change could offset the emissions associated with field GHG emissions and energy inputs (Gan et al., 2014). However, SOC is often not routinely evaluated. Specifically, SOC is often omitted in LCA studies for agricultural systems, and considerations for maintaining paddy C stocks are not frequently considered (Goglio et al., 2015; Liu et al., 2023). Therefore, research is needed to evaluate how different components contributing to net system emissions are interconnected, shedding light on the potential for mitigation practices that may be effective in one component to have unintended consequences for another component. For example, the potential for tradeoffs related to C cycling is particularly unique in anaerobic rice soils, given the high rates of CH_4 emissions but also the strong potential for building SOC.

This systematic review integrates scientific evidence into a comprehensive framework for reducing net system emissions from rice systems in SEA. The first objective was to synthesize information on effective mitigation opportunities for reducing net system emissions focusing on the following four components: (1) field GHG emissions, (2) energy inputs, (3) residue utilization beyond the field, and (4) SOC change. While mitigation opportunities exist within each of the four components, their relative magnitude in terms of CO_2 equivalents is unclear. From reviewing the literature, we also note that a single management strategy can have effects across multiple components (e.g. straw removal can decrease CH_4 emissions but can potentially increase SOC), thus it is important to understand synergies and tradeoffs at the system level. We refer to these interactions as "cross-component effects". The second objective was to explore the cross-component effects of promising mitigation practices to illustrate the fundamental challenges in reducing GHG emissions in one component without adversely impacting other components. Along with that, we identified knowledge gaps in the current literature and prioritized areas for future research using the net system emissions framework.

2. Methods

2.1 Systematic search

We conducted a systematic literature review using the "Scopus" database in June 2020 following established protocols (Koutsos et al., 2019; Moher et al., 2009). The search was performed with combinations of search terms that corresponded to geographical specificity and subject matter interest. The former focuses on SEA and its member nations while the latter focuses on the different components of net system emissions in rice cropping systems (Table 1).

Table 1: Search terms used in the systematic search in Scopus. Geographic specificity refers to geographical locations in SEA. Subject matter interest are divided into the 4 components of net system emissions

Geographic specificity	
Southeast Asia/SEA, Malaysia, Vietnam/Viet Nam, Indonesia, Myanmar/Burma, Singapore, Brunei/Brunei Darussalam, Cambodia, Philippines, Laos/Laos PDR, Thailand	
Subject matter interest	
Field GHG emissions	Greenhouse gas, CH_4, Methane, N_2O, Nitrous Oxide, Climate change, Global warming potential, Emissions
Energy inputs	Carbon footprint, Energy, Life cycle analysis, LCA, Fertilizer, Nitrogen fertilizer, Phosphorus fertilizer, Fossil fuel, Fuel usage, Energy efficiency
Residue utilization beyond the field	Air pollution, Straw, Burn, Straw burning, Residue management, Residue cover
SOC	Soil organic carbon, SOC, Soil carbon, Soil organic matter

These search terms produced a total of 1973 hits (Table 2). Studies that were selected satisfied geographical specificity and subject matter relevancy. Only field-based studies, reviews, or meta-analyses were selected. Opinion papers, greenhouse studies, modeling studies, and studies that were deemed not scientifically rigorous were rejected. To identify mitigation opportunities for each component, studies were selected if they quantified reductions in field

GHG emissions, changes in energy use or GHG emissions (components of energy inputs and residue utilization beyond the field), or SOC change. A total of 1506 records were screened, of which 99 met previously outlined criteria (Table 2). For a full list of papers used in this review, please refer to supplementary materials. Numerical data for variables corresponding with each component was extracted directly from papers if presented in table form. Where results were presented in graphical or figure form, numerical data was extracted using the WebPlotDigitizer (Rohatgi, 2012).

Table 2: Summary table of literature search. For the "Records Retained" row, some studies were used for analysis in more than one component of net system emissions. The total number of studies used remains at 99. For the list of all studies shortlisted, refer to supplementary material.

	Field GHG emissions	Energy inputs	Residue utilization beyond the field	SOC	Total
Hits	564	779	522	108	1973
Number of duplicates			467		
Records Screened			1506		
Records Excluded			1367		
Records Retained	38	33	18	12	99

2. 2 Conceptual framework of net system emissions components

To accurately determine mitigation opportunities within each of the four components contributing to the net system emissions of rice cropping systems, a review protocol should be created to ensure consistency (Moher et al., 2009). Thus, a conceptual framework generalizing sources of C emission and C mitigation (Fig. 2) was developed from a protocol presented by Liu et al. (2016) for net system emissions analysis of rice systems and the review of Lal (2004) on C emissions from farm operations. Pools that are indicated in red are associated with emissions from the system and pools indicated in green are associated with sequestration in the system (Liu et al., 2014). Notably, SOC can take on both positive and negative values, as soils have the potential to sequester C but SOC stocks can also be depleted when managed unsustainably (Paustian et al., 2016).

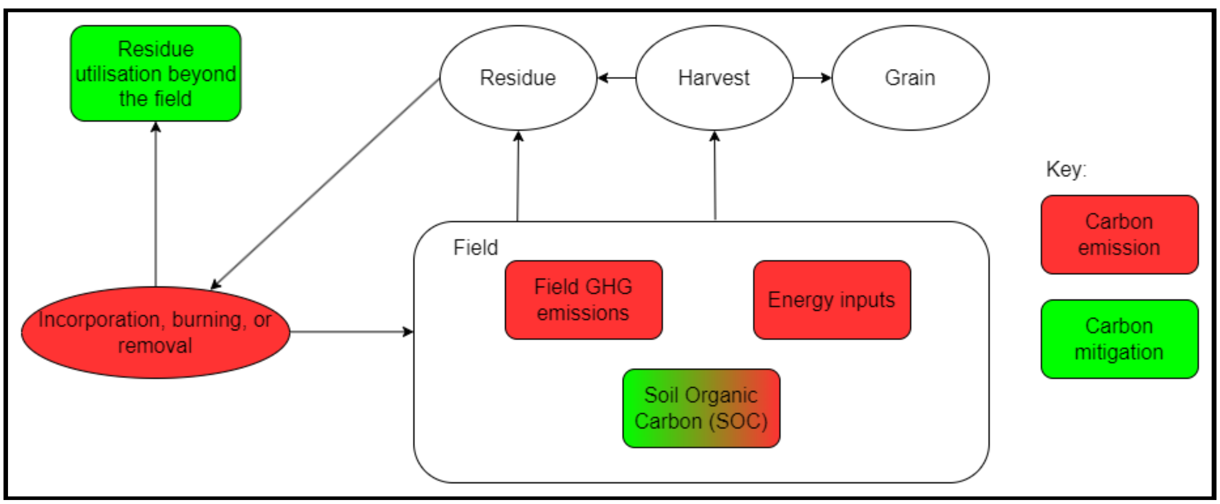


Fig. 1: Schematic of net system emissions conceptual framework guiding the literature search and review (each colored box represents a pool of C flux, with red representing emissions and green representing mitigation).

All pools or fluxes of the net system emissions analysis were converted to kilograms of CO_2 equivalent (kg CO_2 eq) or kilograms of CO_2 equivalent per unit area (kg CO_2 eq ha^{-1}). For field GHG emissions, GWP (global warming potential) values were directly quoted from studies if expressed in CO_2 equivalents. Where only CH_4 and N_2O emissions were reported, GWP was obtained using a radiative forcing potential for each gas: GWP_{CH4} = 34 and GWP_{N2O} = 298 (Myhre et al., 2013). For the energy inputs component, inputs (e.g. fuel use, N fertilizers, pesticides) were converted into CO_2 equivalents using representative conversion factors in SEA (Nguyen et al., 2019). For residue utilization studies, management strategies were evaluated for their GHG mitigation potential relative to the baseline presented in each study. Five studies evaluated emission reductions when residue was removed for energy generation in units of GHG reduction per unit of energy produced (kg CO_2 eq KWh^{-1}). Six other studies on energy generation and other uses were measured in GHG reduction per ton of dry straw (kg CO_2 eq ton dry $straw^{-1}$). Conversions were made as necessary to scale units to a common currency (e.g. from kg CO_2 eq MWh^{-1} to kg CO_2 eq KWh^{-1}). Studies that did not express results in the stipulated format had their key ideas summarised in written form. For SOC studies, results were expressed as SOC change stock over time per unit area (Mg C ha^{-1} $year^{-1}$) scaled to the top 15 cm of the soil. Results for each component were compiled and reported in the results section (see supplementary material for a list of all literature used). The majority of studies evaluated net system emissions changes in only one component, with no study addressing all 4 components of net system emissions.

2.3 Towards net system emissions

Since comprehensive studies addressing multiple components were not available, data limitations prevented us from estimating net system emissions or quantitatively determining how mitigation practices for one component would impact other components. To synthesize the findings of the review and explore the relative importance of different management practices, including their potential cross-component effects and influence on net system emissions, we created three hypothetical scenarios based on the most promising mitigation options. The baseline scenario included conventional flooding for both a dry season (DS) and wet season (WS) crop in SEA using average field GHG emissions from Yagi et al. (2020). A second scenario focused on multiple drainage events to mitigate CH_4 emissions. To reduce labile C substrate causing elevated CH_4 emissions while still building SOC, the third scenario included straw removal with biochar addition as a stable C source. In each scenario, values of emission or mitigation were estimated for each component using area-scaled CO_2 equivalence (kg CO_2 eq ha^{-1}) and additively summed together to reflect net system emissions. For detailed methods and assumptions made in the scenarios, refer to the supplementary materials. As the scenarios are additive and simplistic, they were only performed to provide a sense of the relative magnitude of emissions or mitigation based on available literature for SEA. They are not intended to capture the full complexity of cross-component effects or serve as a quantitative analysis of emission reductions. Instead, we used the results of the scenarios to outline the most important components in tackling net system emissions and highlighted knowledge gaps present in the literature that are pertinent for further investigation.

As our review only includes available literature for this region, we acknowledge the findings may not be representative of all types of rice cropping systems in SEA. This region is diverse in the types of rice cultivation practiced by farmers, including but not limited to different water management practices (rain-fed or irrigated), cropping intensity (single-crop, double-crop, triple-cropped), and level of mechanization and external inputs (ranging from low to high). It is not the intention of this review to account for all of this variability, nor is it feasible to do so considering the available literature. The studies shortlisted in our review consist mainly of irrigated double-cropped systems in the DS and WS, and the conclusions drawn may not be universally applicable. Additionally, this suggests a “norm” in rice research work in the area using the DS/WS double rice crop “model”. Whether to build on this “model” system or to investigate a more diverse system is a decision that experts in SEA can choose to take, and we hope that our review provides good consolidation that forms a basis for informed decision-making.

3. Results

3.1 Field greenhouse gas (GHG) emissions

We identified a total of 38 studies focusing on field GHG emissions from our literature search. Of these, 19 studies were included in Yagi et al. (2020). The majority of the other 19 studies not included in Yagi et al. (2020) were recently published (after 2018). Results from these new studies largely support the main findings of Yagi et al. (2020) – water management (e.g. AWD and mid-season drainage), straw removal and/or burning, and biochar application are promising technical options for mitigating field GHG emissions.

Table 3: Mitigation potential of technical options aimed at reducing GHG field emissions for additional studies not found in Yagi et al. (2020). Options considered are compared against a baseline and mitigation potential is expressed in CO_2 equivalence (kg CO_2 eq ha^{-1}).

Option	Country	DS GHG reductions (kg CO_2 eq ha^{-1})	DS Baseline emissions (kg CO_2 eq ha^{-1})	WS GHG reductions (kg CO_2 eq ha^{-1})	WS Baseline emissions (kg CO_2 eq ha^{-1})	Baseline management and remarks	Reference
Straw removal/burning	Philippines	3796-4932	8671	*study combined DS and WS emissions into an annual value		Straw retention. Treatment tested both removal and burning	Nguyen et al., 2019
	Thailand	915-2078	4265	-318-1158	4758	Straw retention (reported across AWD and continuous flooding)	Maneepitak et al., 2019
	Philippines	1860	3837	200	3422	Straw retention. Values shown only correspond to continuous paddy rice systems	Janz et al., 2019
	Philippines	2001-3143	3891	1046-1491	4132	Straw retention. Treatments tested across straw partial removal, complete removal, and burning	Romasanta et al., 2017
	Vietnam	3990	24859	2367	12892	Straw retention and continuous flooding	Hoang et al., 2019
	Philippines	1877	5193	*study did not investigate option in WS		Straw retention	Samoy-Pascual et al., 2019
Mid season drainage	Vietnam	10535-17759	30100	*study combined DS and WS into an annual value		Continuous flooding	Tariq et al., 2018
Alternate wet-dry (AWD)	Thailand	1094	9422	563	9266	Continuous flooding	Sriphirom et al., 2019
	Thailand	761	3648	1616	5286	Continuous flooding (reported across straw retention, burning and removal)	Maneepitak et al., 2019
	Myanmar	499	1060	1234	1947	Continuous flooding (reported across different rates of manure application)	Win et al., 2020
	Vietnam	6088	24859	4043	12892	Continuous flooding and straw retention (treatment effects reported to AWD depth of -10 cm)	Hoang et al., 2019
	Philippines	-147-3238	2285-5193	*study did not investigate option in WS		Continous flooding with and without straw retention	Samoy-Pascual et al., 2019

	Vietnam	5245	17030	5838	23540	Continous flooding	Tirol-Padre et al., 2018
	Indonesia	4843	1	6413	17861	Continous flooding	Tirol-Padre et al., 2018
	Thailand	86	746	-	1190	Continous flooding	Tirol-Padre et al., 2018
	Philippines	325	2853	-1587	11333	Continous flooding	Tirol-Padre et al., 2018
Fallow drying	Philippines	2660-3181	3314	59-343	483	Continuously flooded. Only GWP values from fallow periods are considered. DS refers to WS to DS transition. WS refers to DS to WS transition	Sander et al., 2018
Biochar application	Thailand	3658	9107	3407	9007	Continuous flooding and no biochar application. Reduction calculated in comparison to continuous flooding and biochar application. Only methane flux was reported.	Sriphirom et al., 2020
Crop rotation	Philippines	2422-3398	4422	246-2003	4246	Continuous rice. Treatments reported across paddy rice - aerobic rice and paddy rice - maize rotations	Janz et al., 2019

Seven studies investigated the effects of water management, including single and multiple drainage events of mid-season drainage and AWD on field GHG emissions (Table 3). Win et al. (2020) also presented novel data from Myanmar, a country that was previously unaccounted for by Yagi et al. (2020). All 7 studies showed that drainage reduced GWP compared to a baseline scenario of continuous flooding (Hoang et al., 2019; Maneepitak et al., 2019; Tariq et al., 2018; Tirol-Padre et al., 2018; Win et al., 2020). This was primarily attributed to reduced CH_4 emissions facilitated by increased oxidizing and aerobic conditions in topsoils that suppress methanogenesis (Sander et al., 2015). In the same studies, drainage caused increased N_2O emissions which have the potential to increase GWP. Despite such a trade-off, the suppression of CH_4 emissions caused a net GWP mitigation effect, ranging between -147 to 6088 kg CO_2 eq ha^{-1}. Yagi et al. (2020) found that multiple drainages resulted in a 31.1% GWP reduction in DS and 24.6% in WS, with large overlapping confidence intervals for both seasons. In the new studies that we found, mitigation practices in DS and WS also had large variability in performance, ranging between 11.4% to 47.1% (mean 23.3%) and 6.1% to 63.4% (mean 25.4%) in GWP respectively. These results also support the conclusion that the practice of multiple drainage can suppress CH_4 emissions, but with high variability in both seasons. At this juncture, we would also like to highlight that multiple drainage, although effective for suppressing CH_4 emissions, can potentially reduce SOC levels. This tradeoff is futher discussed in section 4.4, water management.

New work also highlighted the need for field GHG mitigation during non-growing periods, especially the fallow transition from WS to DS. Under constantly flooded conditions, the WS to DS transition contributed to 26% of GHG emissions during the DS, but this contribution was reduced by 80.3% to 96.0% (2660-3181 kg CO_2 eq ha^{-1}) with soil drying (Sander et al., 2018). In the context of a seasonal value in the DS in this study, drying reduced overall seasonal emissions by at least 69.9%. Such a finding provides evidence that fallow water management has the potential to substantially reduce overall field GHG emissions of rice production.

Other than water management, straw burning and removal were also key mitigation strategies compared to the baseline management of straw retention. Straw removal and burning represent the removal of a source of labile C that can in turn limit CH_4 emissions. Six studies supported the practice of straw removal and burning, having mitigation effects from 915 to 4932 kg CO_2 eq ha^{-1} in the DS, and -318 to 2367 kg CO_2 eq ha^{-1} in the WS. Notably, Romansata et al. (2017) presented novel data on the amount of CH_4 and N_2O emitted during residue burning itself, with emission factors of 10.04 kg CH_4 ha^{-1} (341.4 kg CO_2 eq ha^{-1}) and 0.154 kg N_2O ha^{-1} (45.9 kg CO_2 eq ha^{-1}) respectively. For future studies and policymaking, it will be important to capture these emission factors associated with burning beyond growing season GHG emissions. We would also like to acknowledge that although straw burning is a good technical option to reduce CH_4 emissions in the next growing season, it is a source of atmospheric pollution and its negative impact on human well-being and the environment can be significant (Shyamsundar et al., 2019).

Biochar application was another option investigated. Biochar has been reported to suppress CH_4 emissions primarily by increasing methanotroph abundance, promoting more oxic conditions due to high porosity in its structure, and increasing the availability of electron acceptors in the soil (Nan et al., 2021). Only one study showed that the application of biochar across different water management and fertilization regimes reduced GWP by 40.2% to 37.8% (3658-3407 kg CO_2 eq ha^{-1}) (Sriphirom et al., 2020). Yagi et al. (2020) also identified biochar application as a viable strategy to reduce net GWP by 20%. Although this option has potent mitigation potential, it is less extensively documented, especially given the variability in the quality of biochar that is dependent on the manufacturing process and plant source.

Finally, it should be noted that field GHG emissions and mitigation potentials differed greatly based on study and geography (Table 3). The default IPCC guidelines and emissions factors, while useful, do not have the precision of a well-consolidated national inventory (Tirol-Padre et al., 2018; Vo et al., 2020). To strengthen emission estimation precision in policy-making, more geo-specific consolidation work should be done by research institutions at the national level (e.g. Vo et al., 2020).

3.2 Energy inputs

A total of 33 studies quantified energy inputs or conducted an energy efficiency analysis for rice systems. The majority characterized energy inputs and outputs based on an inventory of management practices, yields, and emission factors, but did not specifically design experiments or report the mitigation effect of different practices. Thus, results are not summarized in a table but findings were consolidated below with a focus on options for reducing energy inputs to mitigate net system emissions. Studies that compiled the total footprint of energy usage pinpointed agrochemicals, especially synthetic N fertilizers (Bautista & Minowa, 2010; Muazu et al., 2015), and usage of fossil fuels for machinery operations, as main sources of C-related emissions (e.g. Arunrat et al., 2016; Soni & Soe, 2016).

Optimal N fertilizer application was identified as a key strategy for reducing energy inputs and is influenced by factors such as soil characteristics, indigenous soil N supply, and variation in crop yield (Devkota et al., 2019). An important takeaway from multiple studies is that growers are over-applying fertilizers in SEA (Huan et al., 2005; Stuart et al., 2018). For example in Thailand, growers were found to be able to maintain yields with a 26% reduction in the usage of synthetic fertilizers (Panpluem et al., 2019). Correspondingly, the most urgent and practical mitigation is to reduce fertilizer (and embodied energy) inputs and sustain yields through site-specific nutrient management (Attanandana et al., 2010; Haefele & Konboon, 2009). Optimal fertilization was also attractive to growers due to financial savings and the ownership they have over such a practice (Arunrat et al., 2018). At the regional or national levels, clear policies and benchmarks for fertilizer use need to be set and considerable resources, training, and institutional support are needed by extension networks for N fertilizer reductions to be realized (Thwe et al., 2019).

Another option investigated was to use other nutrient sources to supplement crop nutrient demand and reduce the use of synthetic fertilizers. Our review identified planting legumes in the previous season (Thwe et al., 2019), residue incorporation (Linquist et al., 2007; Mendoza, 2004), biochar application (Mohammadi et al., 2016, 2017), and other practices (manure, weed biomass, indigenous lime) (Roder et al., 2006) as potential techniques. Organic sources of nutrients need to be mineralized in the soil before they are available for crop uptake, thus the quality of the amendment (e.g. different feedstocks of biochar production, C:N ratio of rice straw or legumes, etc.) and the amount of mineral N that can be supplemented is less predictable and more knowledge-intensive in execution. These methods need to be field tested before they can be reliably implemented. Other techniques that increased nutrient use efficiency such as application of biofertilizer (Banayo et al., 2012), using Azolla cover (De Macale & Vlek, 2004), and type of application method (e.g. surface vs basal) (Sanusan et al., 2009) were also reported to reduce synthetic fertilizer usage. We note that the addition of organic material such as green manure and farmyard manure to reduce N inputs comes with a major tradeoff of increasing CH_4 emissions in the field GHG emissions component (Linquist et al., 2012).

Other agrochemicals, notably the over-application of pesticides in Cambodia, also caused higher emissions (Flor et al., 2019). For the case of fossil fuels, no studies we reviewed assessed tillage intensity and the potential for reductions in fuel consumption associated with reduced tillage. This is an important knowledge gap, as research elsewhere has shown that machinery use and diesel consumption represent a large proportion of total energy consumption, but this can be significantly reduced through changes in tillage (Yadav et al., 2020). Rather, several studies compared the usage of pumps for water reuse compared to surface application with no water reuse (Hafeez et al., 2014; Maraseni et al., 2010). They found water reuse resulted in higher water use efficiency but higher net system emissions due to greater fuel consumption. As such, from a net system emissions perspective, it is recommended that water reuse only take place in areas with water scarcity. Connecting this with AWD, Carrijo et al. (2017) found that AWD can reduce water use by 25.7% compared to continuous flooding. Consequently, by using AWD or other less water-intensive irrigation methods, reduced energy use can likely be attained.

3.3 Residue utilization beyond the field

A total of 17 studies assessed options for residue utilization beyond the field to produce energy or substitute materials used in other agricultural production systems. Eight of these quantified emissions mitigation through straw removal and subsequent electricity generation, bio-DME (dimethyl ether) production, mushroom cultivation, or bioethanol production (Table 4). The range of net GHG reduction was 0.000028 - 1.25 kg CO_2 eq KWh^{-1} or 50.3 – 504.9 kg CO_2 eq per ton of dry straw, as measured in CO_2 equivalence by energy or straw basis. Studies primarily followed a lifecycle analysis (LCA) approach but differed in quantification methodologies and the baseline scenario for evaluating changes in GHG emissions. The majority of studies pointed to the avoidance of straw burning and the substitution of fuel or energy from fossil fuels as sources of emission reductions. While the range of values reported is large due to the use of different LCA inventories and calculation assumptions, all studies consistently showed reductions in emissions if residue was removed for utilization beyond the field.

Interestingly, 2 of these studies showed that even without accounting for the substitution of grid electricity, reductions in emissions can be achieved by avoiding field burning due to reduced CO_2 emissions (Aberilla et al., 2019; Yodkhum et al., 2018). This represents the largest opportunity for reducing net system emissions in this component. However, there is a contention in LCA assumptions specific to avoiding field burning, as some believe the production of CO_2 emissions during field burning can be considered biogenic in LCA assumptions (i.e. the net C balance is considered neutral because CO_2 fixed by the crop is returned to the atmosphere through burning), and should not be considered for net system emissions savings. When examining the mitigation potential of straw removal for the avoidance of burning, the assumptions involved in quantification need to be made transparent and the interpretation of results needs to be contextualized based on the assumptions used.

Table 4: Mitigation potential of utilizing residue beyond the field. The four main options included electricity generation, manufacturing of bio-DME, mushroom cultivation, and bio-ethanol production. Mitigation potential is expressed per unit residue weight (kg CO_2 eq per ton of dry straw) or per unit energy produced (kg CO_2 eq kWh^{-1}).

Option	Country	Net GWP reduction by energy (kg CO_2 eq KWh^{-1})	Net GWP reduction by residue (kg CO_2 eq per ton dry straw)	Sources of emission reductions	Reference
Electricity generation	Thailand	-	116	No straw burning, grid electricity substitution	Silalertruksa and Gheewala, 2013
	Thailand	-	50-216	Grid electricity substitution, reduced field GHG emissions	Jakrawatana et al., 2019
	Thailand	-	375	No straw burning	Yodkhum et al., 2018
	Thailand		447.6	No straw burning, grid electricity substitution	Delivand et al., 2012
	Southeast Asia	0.067- 0.127	-	No straw burning	Aberilla et al., 2019
	Thailand	0.745-0.783	-	No straw burning, grid electricity substitution	Suramaythangkoor and Gheewala, 2011
	Thailand	1.25	-	No straw burning, grid electricity substitution	Suramaythangkoor and Gheewala, 2008
bio-DME	Thailand	-	245	No straw burning, LPG substitution	Silalertruksa and Gheewala, 2013
	Thailand	0.000028	-	No straw burning, LPG supplement	Silalertruksa et al., 2013
	Thailand	0.003	-	No straw burning, fuel for diesel engine	Silalertruksa et al., 2013
Mushroom cultivation	Vietnam	-	102.8	No straw burning	Arai et al., 2015
bio-ethanol	Thailand	-	283	No straw burning, gasoline substitution	Silalertruksa and Gheewala, 2013
	Thailand	-	504.9	No straw burning, gasoline substitution	Delivand et al., 2012

The 9 other studies investigated similar uses of residue but were not included in Table 4 due to different units of measurement (e.g. energy balances instead of CO_2 equivalence). Results supported the finding that residue utilization beyond the field for energy production or as a substitute for bedding materials in mushroom cultivation can mitigate GHG emissions compared to in-field residue burning. From research conducted in Thailand and the Philippines, sizeable emission savings could be achieved if residue was removed from rice fields at the national level (1.81% and 4.31%, respectively of the whole nation's estimated CO_2 emissions) (Gadde et al., 2009).

As discussed earlier, fuel use associated with machinery is an important source of direct CO_2 emissions. Therefore, the mechanical harvesting of residues could potentially offset the benefits of residue removal for energy production. Studies that focused on the quantification of net system emissions of field residue collection showed that even when residues were collected by mechanical means, a small net mitigation effect was attained if the residue was removed for energy production (Balingbing et al., 2020; Nguyen et al., 2016b). For example, each ton of straw required 70–160 kg CO_2 eq for collection, but subsequent power generation created a net mitigation of 87 kg CO_2 eq per ton of straw from fossil fuel substitution (Nguyen et al., 2016b). Similarly, energy balances were also positive (e.g. use of rice straw for biogas production creates a positive net energy balance of between 70% and 80%), after accounting for energy inputs to grow rice and harvest residues (Nguyen et al., 2016a). Other studies also showed that residue utilization generated more energy than was used for the cultivation of rice (Lecksiwilai et al., 2015; Nguyen et al., 2016a). Such findings support the mechanical collection of residues, especially given decreasing labor availability in the region (Nguyen et al., 2016b). In this section, we focused on the straw fraction of residue, but there is also the potential for generating energy using rice hulls that are a waste product of the milling process (Mai Thao et al., 2012; Prasara-A & Grant, 2011). This was outside the scope of our study because it does not influence field management practices and could apply to any rice crop being harvested and milled.

3.4 Soil organic carbon (SOC) change

Thirteen studies measured SOC change due to field management practices. The results of 11 studies are compiled in Table 5 in terms of the rate of SOC change (Mg C ha^{-1} $year^{-1}$). These studies identified a variety of options for building SOC, ranging from long-term rice cultivation with flooded soils to higher C inputs (e.g. biochar, straw, or compost) to no-till and crop rotation practices. Research in SEA has documented that SOC in paddy systems can increase with continuous rice cropping. For instance, the SOC density of topsoils in Java increased from 7.6 g kg^{-1} to 11.7 g kg^{-1} from 1980–1990 to 1990–2010 (Minasny et al., 2012). Arunrat & Pumijumnong (2017) found a similar range of SOC increase over 10 years in Thailand, although measurements were made at 0-40 cm rather than 0-15 cm soil depth.

To further enhance the accumulation of SOC in paddy systems, the majority of studies evaluated the application of external amendments containing large amounts of C, most notably biochar and rice straw. Biochar is a stable form of organic C made up of recalcitrant compounds such as lignin that are resistant to microbial decomposition (Kuzyakov et al., 2014; Marschner et al., 2008), making it particularly effective for increasing SOC. In Thailand, biochar application at very high rates (6.25 Mg ha^{-1} to 25 Mg ha^{-1} per growing season) in a single-season experiment increased SOC by 3.74 to 26.74 Mg C ha^{-1} $year^{-1}$ (Thammasom et al., 2016). Another study reported results on the lower end of this range, with biochar (application rate 10 Mg ha^{-1} per season) increasing SOC by 3.64 - 4.74 Mg C ha^{-1} $year^{-1}$ (Sriphirom et al., 2020). While there has been a positive correlation between the rate of biochar application and SOC increase, a consensus for an optimum rate is yet to be found in the region, and research tended to use high rates that may not be practical or economically feasible. High rates of biochar application (>40 Mg ha^{-1}) can potentially increase SOC to a greater degree. However, toxic compounds from the production of biochar and other parameters such as bulk density and nutrient availability could be negatively impacted with overapplication (Gao et al., 2019; Mukherjee & Lal, 2013).

Table 5: Yearly soil organic carbon (SOC) of the compiled studies and the corresponding management option. Increases in SOC are scaled to the top 15 cm of the soil.

Option	Country	SOC change (Mg C ha^{-1} year^{-1})		Duration (years)	Citation
		Mean	Range		
Long-term data (no specific treatment tested)	Indonesia	0.32	-	20	Minasny et al., 2012
	Thailand	0.19	0.86 - 2.71	10	Arunrat and Pumijumnong, 2017
Biochar application	Thailand	10.89	3.74 - 26.74	0.5	Thammasom et al., 2016
	Thailand	2.17	3.64 - 4.74	1	Sriphirom et al., 2020
Rice straw (RS) incorporation	Thailand	-3.87	(-)1.84 - (-)5.12	0.5	Thammasom et al., 2016
	Thailand	0.92	-	10	Oechaiyaphum et al., 2020
	Thailand	0.92	0.31 to 1.38	1	Vityakow et al., 2000
	Philippines	0.11	-0.05 - 0.33	15	Pampolino et al., 2008
Compost amendment	Vietnam	4.03	0.72 - 6.05	13	Watanabe et al., 2017
	Thailand	0.63	-	10	Oechaiyaphum et al., 2020
Green manuring	Thailand	0.18	-	10	Oechaiyaphum et al., 2020
No-till	Cambodia	1.93	1.65-2.37	4	Hok et al., 2015
Crop rotation	Vietnam	0.61	0.14 - 0.97	8	Linh et al., 2015

Rice straw incorporation is also a potential driver of SOC accumulation based on multiple field studies in our review (Oechaiyaphum et al., 2020; Pampolino et al., 2008; Vityakon et al., 2000). Both long-term studies showed consistent increases of 0.92 Mg C ha^{-1} year^{-1} (Oechaiyaphum et al., 2020), and 0.11 Mg C ha^{-1} year^{-1} (3 out of 4 sites reported SOC gains) (Pampolino et al., 2008). Although most studies examined here reported gains in SOC, Thammasom et al. (2016) reported reductions in SOC with rice straw incorporation, which was attributed to increased GHG flux and C mineralization. However, the duration of their study was

0.5 years compared to other long-term studies of 15 years and 10 years by Pampolino et al. (2008) and Oechaiyaphum et al. (2020), respectively, and is not representative of the long-term effects of straw incorporation on SOC change.

Other strategies identified in our review for increasing SOC include compost addition (Oechaiyaphum et al., 2020; Watanabe et al., 2009, 2017), no-till (Hok et al., 2015), crop rotations (Linh et al., 2015) and green manure application (Oechaiyaphum et al., 2020). Another study also related SOC levels to microbial activity, suggesting that reduced pesticide application is key to ensuring microbial activity that can promote SOC accumulation (Maneepitak & Cochard, 2014). Interestingly, one study found that both conventional-till and no-till can increase SOC over time, although this was to a smaller extent for conventional-till compared to no-till (Hok et al., 2015), supporting the long-term trend of net SOC gain in paddy systems presented by Minasny et al. (2012). As such, even when research experiments do not include treatment or practice designed to specifically build SOC, these results suggest that studies on net system emissions should always consider the effect of SOC change in rice systems. If there is a trend of SOC gain as observed in this review - this would improve the overall net system emissions of rice by offsetting emissions in the other components (Jat et al., 2022).

4. Synthesis and Future Directions

4.1 Overview

Generally, two effective mitigation strategies for reducing net system emissions stood out. The first was to directly manipulate C availability through C removal (e.g. straw removal), which can limit SOC gain but also reduce CH_4 emissions (Tables 3 and 5). The second was to indirectly manipulate C cycling by introducing non-flooded periods, with aerobic soil conditions suppressing methanogenesis (Table 3). Multiple studies showed that adding C inputs to the field (e.g. crop residues, organic amendments, or biochar addition) was effective at building SOC (Table 5). Water management such as single and multiple drainage events was found to effectively reduce field GHG emissions, as well as biochar addition and straw removal. Also, GHG mitigation can be achieved by optimizing energy inputs, specifically N fertilizers. Residue utilization beyond the field helped avoid emissions from open-field burning, produced energy that offset emissions associated with conventional fuel and energy sources, or substitute material used in other production systems (Table 4).

The results identified promising mitigation opportunities within each component, but the magnitude of net system emissions mitigation is more complex due to cross-component effects. Within each component, it is comparatively easy to predict how a specific practice can alter GHG emissions as there are factors that strongly affect its magnitude (e.g. amount of C input for SOC and weight of straw used for electricity generation in residue utilization). However, C cycling pathways in flooded rice soils are not independent and mitigation practices can have consequences across multiple components. Yet it was rarely acknowledged in studies that effective practices for one component may cause important synergies or tradeoffs for another component. For example, straw removal can mitigate CH_4 emissions and its subsequent utilization can produce energy, but limits the potential to build SOC due to lower C input into the soil. Similarly, utilizing organic sources of nutrients such as green manure can reduce energy inputs through reduced application of synthetic fertilizers, but will likely result in greater field GHG emissions from CH_4, while also having the potential to increase SOC. Despite these cross-component effects, no empirical study shortlisted from our review examined all 4 components, representing a knowledge gap. In the face of climate change, it is prudent to explore potential cross-component effects and tradeoffs to understand how different mitigation practices affect net system emissions. With this in mind, this section discusses the current knowledge of cross-component effects based on simplified scenarios constructed from the results above. The goal is to outline the most important knowledge gaps currently present and explain the need for a shift towards net system emissions accounting to resolve key challenges in this field of study.

4.2 Magnitude of emissions in each component

A climate-smart rice management system in the framework of our 4 components will have (1) mitigated field GHG emissions, (2) reduced energy inputs, (3) efficient utilization of residue beyond the field, and (4) an increase in SOC. We generated three scenarios to explore the feasibility of simultaneously reaching these outcomes and understand which components affected net system emissions the most. The scenarios included (a) a baseline, (b) multiple drainage, and (c) straw removal plus biochar application to illustrate cross-component effects and the relative impact of each component on net system emissions (Table 6, see supplementary material for calculations). The baseline scenario assumed continuous flooding, high energy inputs, and no specific straw management or residue management. We acknowledge that research-wise, continuous flooding is a commonly used baseline in Southeast Asia (Yagi et al., 2020), but what is practiced by growers may differ. The multiple drainage scenario used multiple drainage compared to the baseline scenario. Lastly, the straw removal plus biochar application scenario had the management practices of straw removal and utilization, and biochar application. The energy and emissions associated with biochar production were not included in our analysis. In each scenario, the magnitude of each component (i.e. field GHG emissions, input energy, residue utilization, and change in SOC) was calculated based on the management practices and summed to provide an estimate of net system emissions.

Table 6: Relative contribution of four components to net system emissions. Scenarios reflect promising mitigation practices for either water or C management identified in the review. Emissions are presented in CO_2 equivalents ha^{-1} yr^{-1} which includes 1 DS and 1 WS.

All values in the unit of kg CO2 eq ha^{-1} yr^{-1}	Conventional	Multiple drainage	Straw Removal/Utilization Biochar application
Field GHG emissions	23665	17331	13064
Energy inputs	2060	2060	1699
Residue utilization	0	0	-354
SOC change	-1173	1351	-10105
Net system emissions	24552	20741	4305

A key finding of the scenarios we generated was that field GHG emissions and SOC change were the most important components responsible for net system emissions, representing the largest source of emissions and mitigation, respectively, compared to energy inputs or residue utilization beyond the field (Table 6). For the field GHG emissions component, an average baseline emissions value of 23665 kg CO_2 eq ha^{-1} yr^{-1} was obtained with continuous flooding as the water management strategy. Field GHG emissions was a key component in reducing net system emissions, with multiple drainage reducing emissions by 6335 kg CO_2 eq ha^{-1} yr^{-1}, and straw removal/utilization and biochar application reducing emissions by 10601 kg CO_2 eq ha^{-1} yr^{-1}. SOC change was another key component. By adding biochar, SOC gain can mitigate emissions by -10105 kg CO_2 eq ha^{-1} yr^{-1}, although this value has high uncertainty (red highlight).

Comparatively, the other two components of energy input and residue utilization provide smaller mitigation. The energy input component of a baseline scenario with a high degree of mechanization and high levels of synthetic inputs (fertilizers and pesticides) totaled only 2060 kg

CO_2 eq ha^{-1} yr^{-1}, or less than 10% of net system emissions. If straw was removed, the residue utilization component provides mitigation at -354 CO_2 eq ha^{-1} yr^{-1}, roughly around 1% of net emissions (only accounted for mitigation from power generation and not prevention of straw burning). From the relative magnitude of emissions presented in these scenarios, it suggests that preliminarily, field GHG emissions and SOC are the most important components to tackle to achieve optimum net system emissions.

Importantly, we found the components of energy input and residue utilization to be largely standalone components with mitigation easily accounted for (highlighted in green). Energy input is mostly improved through reducing inputs (e.g. reduced fertilization with optimum rates, reducing machinery use with direct seeding etc.) and mitigation potential is easily verifiable when the analysis accounts for energy savings. For residue utilization, mitigation can be effectively calculated with the amount of straw utilized for C mitigating purposes (the benefit of reduced CH_4 is accounted for in the field GHG component). Both components of energy inputs and residue utilization can be quantified using LCA methodologies and are in the authors' opinion, easy to account for given the specific management practices at a local level based on the breadth and quality of studies we shortlisted through this review. More importantly, from a C cycling perspective, the challenges of cross-component effects must be accounted for within field GHG emissions and SOC components, whereas energy inputs and residue utilization components are somewhat independent.

We illustrate the ease of accounting in the energy input and residue utilization components with two examples. First, results show that the input of organic amendment (e.g. biochar) can potentially replace synthetic fertilizer to meet crop N demand, reducing energy input, while also altering SOC and field GHG emissions. The mitigation effect of organic amendments in the energy input component through reduced N fertilizer input can be easily accounted for using LCA inventories. Comparatively, organic amendment input likely has a cross-component effect of increasing field GHG emissions and SOC gain which is considerably more difficult to estimate. An example that shows the ease of accounting for residue utilization is the practice of straw removal. Straw removal in our scenario reduced field GHG emissions but may impede building SOC, making the cross-component effect of these two components difficult to estimate. However, if the removed straw was utilized for electricity generation or other C mitigating practices, it is an "add-on" that is easily accounted for using LCA in the residue utilization component based on how much straw was removed and what it was used for.

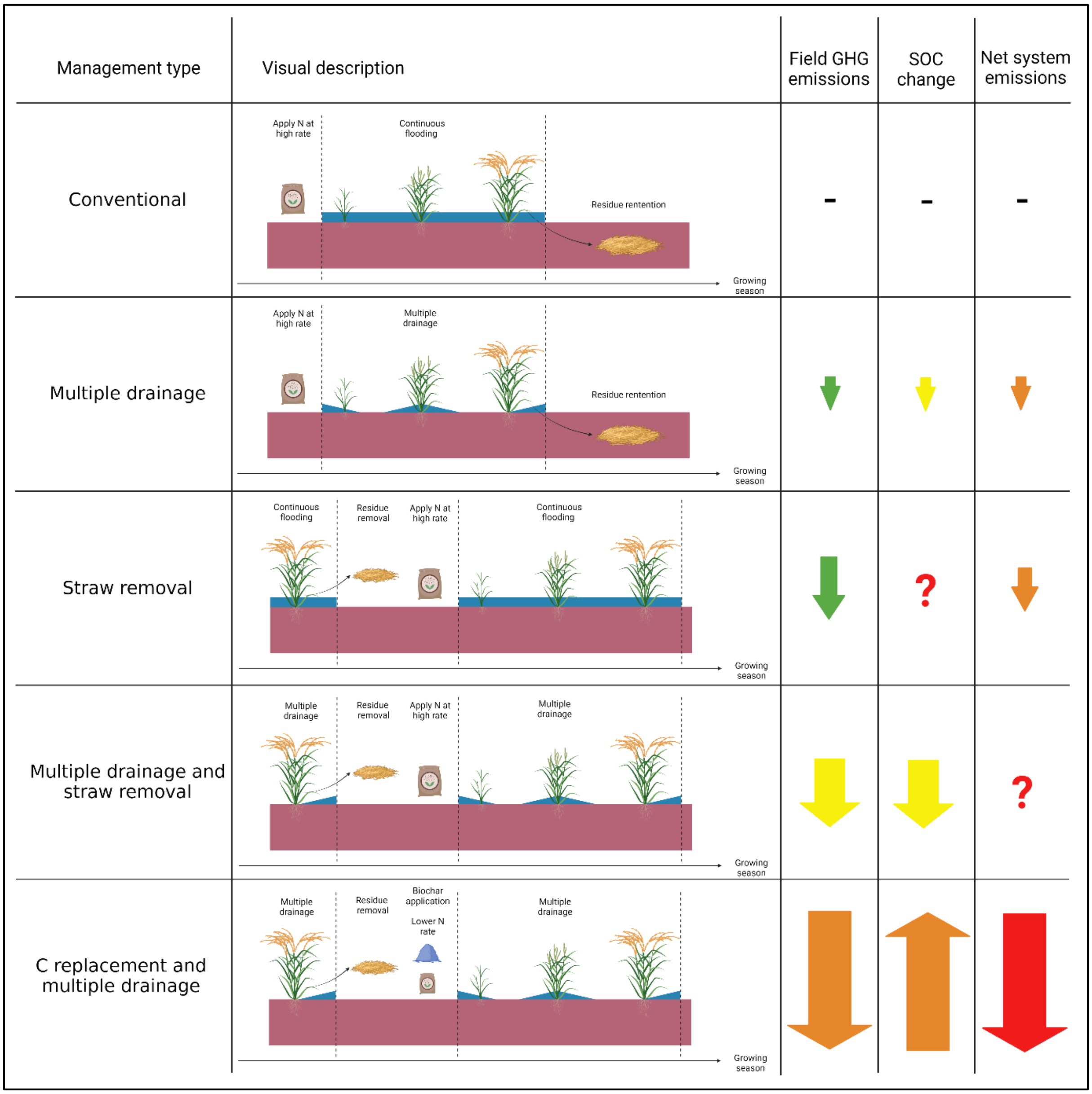
Having said that, each LCA study utilizes different LCA inventories and assumptions. In the residue utilization component of our scenario, we did not account for the CO_2 savings from the prevention of straw burning. If accounted for, the mitigation level can potentially increase to more than -1700 kg CO2 eq ha^{-1} (mitigation value calculated with results from Delivand et al., 2012). We would like to highlight that relative magnitudes of mitigation can shift considerably depending on boundaries and assumptions (Table 4, Sources of emission reductions), specifically (1) CO_2 savings from no straw burning, and (2) its subsequent usage for mitigation such as power generation. If assumptions and boundaries are standardized, and common inventories are established for LCA accounting in components of energy input and residue utilization at the regional level, they will be components that are comparatively easier to account for in net system emissions. Consequently, the rest of the "Synthesis and Future Directions" section will focus on the two components of field GHG emissions and SOC change, focusing on their associated potential for mitigating net system emissions, and specific knowledge gaps that need to be investigated.

4.3 Uncertainties for field GHG emissions, SOC change, and net system emissions

The construction of our scenarios allowed us to preliminarily understand (1) the main management strategies (water management and C inputs) that reduce net system emissions, (2) the main components to target (field GHG emissions and SOC change) that contribute to net system emissions mitigation, and (3) the knowledge gaps that made the estimation of cross-components effects in these two main components difficult. The results and takeaways from our scenarios and available literature were used to create Fig. 2, where we assessed the relative impact of best management practices (multiple drainage, straw removal, multiple drainage and straw removal, and C replacement with multiple drainage) on field GHG emissions, SOC, and net system emissions. This assessment included magnitude, direction, and confidence level.

Figure 2: Conceptual figure summarizing the relative impact of 4 mitigation strategies (multiple drainage, straw removal, multiple drainage and straw removal, and C replacement and multiple drainage) on the components of field GHG emissions, SOC change, and net system emissions compared to a conventional baseline. A visual description of the scenario is shown together with arrows that show the approximate magnitude (size of arrow), likely directionality, and confidence level of its impact. For visual descriptions with a previous season, it is done so to highlight residue management impacts on emissions in the next season. Downward pointing arrows suggest a decrease in field GHG emissions, a decrease in SOC, and a decrease in net system emissions. The confidence level is shown through color: Green (confident), light yellow (somewhat confident),

orange (somewhat confident but with little data supporting), red (somewhat confident but no empirical verification). A question mark shows knowledge gaps large enough that no conclusions can be drawn.



On a broad level, the relationship between field GHG emissions and SOC change will determine whether it is possible to reduce net system emissions. Key challenges related to water management and C inputs are discussed below. Reducing field GHG emissions through water or straw management are the largest and most well-studied opportunities as indicated by the green arrows in Fig. 2. However, there is little research assessing how water and straw management may alter SOC for rice systems in SEA. Consequently, there is reduced confidence in SOC change for multiple drainage and straw removal. Importantly, there is no information on SOC change for straw removal (Fig. 2). Considering that soil C cycling is dependent on microbial-mediated processes over long temporal scales, and C sources can come from various sources such as rice straw, rhizodeposits, and rice roots, it is critical to understand how straw removal affects SOC levels in the long term (Liu et al., 2019).

The co-adoption of multiple drainage and straw removal would likely result in enhanced mitigation for field GHG emissions but has not been studied, resulting in its effects having a lower confidence level compared to the two managements being used independently (Fig. 2). Given the higher potential for SOC losses with multiple drainage and straw removal, a key knowledge gap is whether this would increase net system emissions. To specifically manage for SOC increase, we propose the idea of C replacement – removal of labile C (straw) and addition of more recalcitrant C (we used biochar as an example). When C replacement is "stacked" with multiple drainage, this combination could potentially mitigate field GHG emissions and build SOC at the same time (Sriphirom et al., 2020). The "stacking" of C replacement and multiple drainage is a theoretical best that provides optimal mitigation to net system emissions, but has low confidence due to insufficient empirical verification through long-term experiments. Moving forward, we argue that research evaluating practices to reduce CH_4 emissions should also ensure that SOC losses do not occur, as this could offset the climate benefits. At the same time, if SOC can be increased through C replacement (e.g. biochar application) while mitigating field GHG emissions, this would represent a theoretical best for reducing net system emissions from a technical standpoint.

4.4 Water management

From the results of our review, it is clear that single and multiple drainage events such as AWD simultaneously achieve comparable yields to continuous flooding and reduced emissions if implemented well (Carrijo et al., 2017). What is not as clear, is if the introduction of aerobic soil conditions can cause a decrease in SOC (Fig. 2), and if so to what quantitative degree (Fig. 2, managements with multiple drainage)? Due to flooded conditions in rice paddies, anaerobic conditions are present in soils extensively during the growing season, leading to slower rates of organic matter decomposition compared to aerobic microbial respiration in non-flooded soils (Pan et al., 2010; Sahrawat, 2012), allowing for greater stabilization of SOC. Hence there is concern that non-flooded soil conditions designed to mitigate CH_4 emissions may increase aerobic microbial C respiration and decrease SOC that would otherwise be retained in the system. Preliminary work from 12 studies in other regions showed an increase in soil CO_2 emissions and a corresponding decrease in SOC with drainage that introduces aerobic soil conditions (Livsey et al., 2019). Similarly, Shang et al. (2021) found that AWD did not result in a net GHG benefit due to SOC losses being higher than the reduction in CH_4 emissions. In contrast, another study showed that AWD does not reduce SOC for 3 years (Tirol-Padre et al., 2018). Overall uncertainty exists between drainage and SOC change in terms of net system emissions, because a reduction in CH_4 emissions is a short-term C flux while the SOC change is a long-term process. Furthermore, SOC will neither increase nor decrease infinitely, with SOC changes generally occurring over long temporal scales until a new equilibrium stage is reached between C sequestration and soil respiration. However, this potential tradeoff is an important knowledge gap that requires further research, especially due to long-term implications for soil fertility and climate change mitigation (Livsey et al., 2019). The implications for net system emissions are further complicated given large variation in the intensity of AWD implementation as factors such as frequency, drain duration, and soil moisture levels can all contribute to SOC mineralization. Loss of SOC will probably be greatest in systems where AWD or intermittent irrigation is done many times during the season. Comparatively, one to two drainage events during the season may be able to achieve large CH_4 reductions without reducing SOC, representing a sweet spot and is already commonly practiced in some parts of the world such as China and California (Perry et al., 2022; Wang et al., 2020). However, the ability to one to two drainage events reduce CH_4 while maintaining SOC requires empirical verification.

In theory, drainage should be compatible with other GHG mitigation practices identified in this review, such as residue removal. One documented tradeoff in using drainage is the increase of N_2O emissions when a high concentration of soil N is present (Linquist et al. 2012). Despite the increase in N_2O emissions, the large reduction in CH_4 emissions generally creates a net mitigation effect in field GHG emissions (Yagi et al., 2020). The co-adoption of residue removal with multiple drainage should have cross-component benefits as it supports both the mitigation opportunities for residue utilization beyond the field (Table 4) and reduced field GHG emissions (Fig. 2, multiple drainage and straw removal). The effect for GHG mitigation should be greater for the “multiple drainage and straw removal” management as it decreases C available for methanogenesis and promotes aerobic soil environments. However, this specific practice potentially reduces SOC as it actively prevents C inputs into the system and promotes aerobic respiration. Consequently, it is difficult to draw any conclusions on net system emissions when "stacking" residue removal with multiple drainage practices.

The key consideration is that a reduction in field GHG emissions is categorized as "avoidance" of emissions while building SOC falls under the category of "sequestration". While avoidance is per se permanent, sequestration could be reversed (e.g. with a change in crop management). As such, it is important to be aware of this relationship, especially in the case of drainage. Building SOC should have an upper limit in its potential for net system emissions, but drainage can provide yearly CH_4 reductions effectively. There are several research questions to address in future work – is field GHG emission mitigation intense enough that it has a larger effect than SOC loss, and how does this relationship change in the long term with varying intensities of drainage management (Fig. 2)? Since drainage and straw removal are both individual potent strategies to mitigate field GHG emissions, their effects of their co-implementation on SOC and net system emissions is a key knowledge gap that warrants more investigation.

4.5 C Management and Replacement

Maintaining sufficient C inputs into rice systems to maintain soil fertility while minimizing CH_4 emissions is a conundrum. The input of residue, manure, and in general, any form of organic amendments that adds to the labile C pool, leads to an increase in field GHG emissions when there is no change in water management (Haque et al., 2020; Azeem Tariq et al., 2017). Rice straw has high concentrations of cellulose, a pool of labile C, that is readily accessible to microbes and can be broken down by microbial action (Puttaso et al., 2011). As such, applied rice straw serves to increase the labile C pool more so than stable C (Yin et al., 2014), making increases in SOC less predictable despite high C input and increased CH_4 emissions.

The direct approach to reducing CH_4 emissions derived from the soil C pool is to reduce labile C inputs, specifically residue removal (Fig 2, managements with straw removal). While this does not necessarily decrease SOC (Pampolino et al., 2008), it is not well investigated and may limit increases in SOC over the long term. By extension, the effects of residue removal on net system emissions are unclear. From a practical standpoint, residue management might be influenced more by policy and economic viability. For example, rice straw can be removed and used as cattle feed in SEA, representing an income source (Sarnklong et al., 2010). In comparison to Californian rice systems, rice straw incorporation is a default management strategy as burning is highly restricted (Hill et al., 2006; Linquist et al., 2006). Given the variation in regional legislation on residue management, managing for reduced C inputs with straw removal may not be an option for some growers.

From our results, a potential way around the dilemma of achieving high SOC gain with reduced CH_4 emissions, is to substitute labile C with recalcitrant C using external material such as biochar (Fig. 2, C replacement management). Research has shown that C replacement with biochar (or other forms of more stabilized C), representing a pool of recalcitrant C, is less available for microbial action, reducing CH_4 emissions (Haefele et al., 2011). In the case of residue removal, biochar application can replace the removed C source, increasing the potential of the system to gain SOC and potentially providing the largest net system emissions reductions (Fig. 2, C replacement and multiple drainage). Importantly, biochar is an external substrate for rice paddy, and its production is associated with energy inputs, representing a "relocation" of emissions.

Generally, biochar application to soils has a net mitigation effect when SOC increases are accounted for in LCA studies (Matuštík et al., 2020). For example, a C abatement of 0.7–1.3t CO_2 eq per oven dry tonne of feedstock can be achieved depending on the feedstock type (Hammond et al., 2011). If practiced on a large scale, biochar will likely have to be sourced externally outside of rice systems and represent a barrier to implementation.

To further tighten C cycling, removed residue can potentially be utilized for manufacturing biochar during power generation processes (Yaashikaa et al., 2020). If biochar is reapplied to the field, this creates a tight C-cycling loop that can serve as an alternative to direct residue incorporation (Jakrawatana et al., 2019). However, net life-cycle emissions must be determined while considering alternative end uses of the biomass used to create biochar (e.g. rice husk), specifically whether greater mitigation is possible if the residue is used to produce other energy or fuel which offsets fossil fuel use (Paustian et al., 2016).

The long-term stability of materials used for C replacement must also be addressed. Although recalcitrant C represents a pool of SOC that potentially decreases CH_4 emissions, the quality of C can change through time, especially with increased aeration that can transform the soil C pool into labile material from initially recalcitrant material. This raises important questions about how long can SOC be increased for, and if C replacement with more stable material will have a sustained suppression effect on reducing CH_4 emissions and increasing SOC. The most immediate action for research is to conduct more long-term experiments (5-10 years) to answer these research questions to assess changes in C quality, and quantify the degree of SOC gain and its interactions with field GHG emissions, soil fertility, and yields. Although SOC gains hold promising potential, the data present in published literature is of inadequate quality as the majority are short-term studies. More work needs to be done to ensure greater confidence in future net system emissions benefits.

The "stacking" of mitigation practices, especially "multiple drainage and C replacement" (Fig. 2) is an interesting concept to explore as it has the potential to provide mitigation in all 4 components in our framework through (1) reduced field GHG emissions through drainage and stabilized C inputs, (2) reduced N fertilizer input with amendments, (3) removed residue for power generation, and (4) application of stable C to increase SOC. Selected studies have tested several combination practices, with Sriphirom et al., (2020) showing that AWD plus biochar application was effective at mitigating field GHG emissions and building SOC in the short term. However, to our knowledge, this theoretical best concept has yet to be tested under field conditions in long-term experiments.

4.6 Challenges to Implementation and Limitations

While solutions presented in the synthesis have the technical potential to reduce net system emissions, we would also like to acknowledge challenges and limitations from stakeholder and implementation standpoints. Multiple drainage or AWD are the most immediate solutions for reducing field GHG emissions, but infrastructural inadequacy in the region can pose uncertainty over drainage and flooding events (Enriquez et al., 2021; Quang et al., 2019). The reader is referred to Enriquez et al. (2021) for a full discussion of the successes and challenges associated with scaling up AWD.

Residue utilization beyond the field similarly has barriers to implementation at the field and industry levels. At the field level, the tight timeline in double or triple-cropped rice systems between harvest and land preparation poses considerable difficulty. The cost of labor for harvesting residue also means that it is currently not profitable (Wassman, 2019). At the industry level, biomass availability is seasonal (Cheewaphongphan et al., 2018), with rice residues mainly available at the middle and end of the year. For power plants or facilities that process residue, other sources of biomass must be used during other parts of the year to ensure continuality and viability in operations, making this an endeavor requiring collaboration across multiple agricultural sectors or cropping systems (Cheewaphongphan et al., 2018; Tun et al., 2019). However, with the increasing use of straw balers in smallholder rice systems, residue transportation and storage get easier (Kumar et al., 2023). Viable alternatives for commercial use of rice residues are not widely available and further investment and enabling policies are required to make the practice of residue removal economically feasible to achieve GHG reduction benefits.

Biochar stands out relative to other strategies in terms of its mitigation potential, especially if derived from collected rice residues, but barriers to implementation and knowledge gaps need to be addressed (Guo et al., 2015). Additionally, biochar can also come from different feedstocks, resulting in materials that differ in quality. Furthermore, biochar must be produced under controlled conditions to avoid the development of toxic substances (Shi et al., 2023). The effect of using different types of biochar and associated rates also needs better quantified before large-scale adoption (Awad et al., 2018).

5. Conclusions

Our review highlighted opportunities for net systems GHG mitigation by considering field GHG emissions, energy inputs, residue utilization beyond the field, and SOC change. Given that SEA is a key producer and exporter of rice in the global economy, there is increasing emphasis from governments and the private sector to meet international climate change commitments by reducing net system emissions from rice systems. Our integrated analysis brings attention to other components of net system emissions beyond field GHG emissions, especially relationships between CH_4 mitigation and SOC that have not been considered in previous work in this region. Results show SOC is suggested to be at least maintained, if not slightly increased, with continuous rice cropping and is a mitigating factor of net system emissions in most cases. Thus, studies of net system emissions in rice systems should strive to include SOC change as part of their calculations. Other findings from our review support the consensus that field GHG emissions are the main source of net system emissions and can be controlled by well-established strategies, especially water management such as AWD. Energy inputs and residue utilization beyond the field represent a small source of emissions and likely do not affect net system emissions extensively in SEA rice systems. Overall, we have two main takeaways. First, multiple drainage and C removal can reduce CH_4 emissions, but can negatively affect SOC. Second, as field GHG emissions and SOC change are the largest contributing components to net system emissions, they need to be considered for effective climate change mitigation.

From our review, we propose 3 broad questions for future research – (1) what is the tradeoff between reduced CH_4 emissions and potential SOC loss when using drainage or reducing C inputs, (2) what long-term effects does C replacement (using more stable forms of C) have for the various components of net system emissions, and (3) does the “stacking” of best management strategies (especially “multiple drainage and straw removal” and “C replacement and multiple drainage”) have a net mitigation effect for net system emissions? Future research, especially long-term research, is necessary to test combinations of strategies under field conditions across long temporal scales to fully understand the optimal point between the largest emitter and sequestrator – field GHG emissions and SOC. The value of our findings lies in the potential that an optimally managed system can be more sustainable than previously thought, if not at least capable of feeding a growing world population while being gentler on our planet.

Conflict of Interest

All authors declare no conflicts of interest.

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Chapter Two

Title:

Introduction of a fallow year to continuous rice systems enhances crop soil nitrogen uptake

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Abstract

Rice grown in California constitutes 20% of total U.S. rice production and is typically grown in a continuous rice monoculture system. In recent years, growers have been forced to fallow their lands often due to winter droughts leading to water restrictions or spring rains leading to prevented planting. Increased soil aeration due to fallowing creates knowledge gaps in soil nitrogen (N) availability. A two-year field study was conducted to evaluate differences in crop N uptake between rice cultivation following a fallow season, fallow rice (FR), and continuous rice (CR) systems. Crop uptake of soil N (N $uptake_{soil}$) and fertilizer N (N $uptake_{fertilizer}$) were quantified using ^{15}N-enriched ammonium sulfate applied in microplots as a preplant (150 kg N ha^{-1}) or topdress (30 kg N ha^{-1}) application. In both seasons when N was applied as a preplant fertilizer, the FR treatment had higher grain yield than did the CR treatment, with yield differences of 2.3 Mg ha^{-1} in 2021 ($P<0.05$) and 1.7 Mg ha^{-1} in 2022 ($P<0.05$). Examining the sources of crop N uptake for preplant applied N, on average, N $uptake_{soil}$ in the FR treatment was 16.7 kg N ha^{-1} higher than the CR treatment at maturity ($P<0.05$). In contrast, N $uptake_{fertilizer}$ was similar between treatments. Additionally, comparable soil and crop fertilizer N recoveries in CR and FR preplant N suggested that the pathways and magnitudes of fertilizer N losses were similar in both systems. These results indicate that N $uptake_{soil}$ was primarily responsible for lower N uptake in CR. Similar results were found when N was applied as a topdress, where FR had increased N $uptake_{soil}$ in both years. We further investigated the reason for lower rates of N $uptake_{soil}$ in CR. Soil phenols, which have been documented to accumulate in continuously flooded rice systems and stabilize soil N, were quantified in the field study. Complementing the rigorous field study, a regional survey study that incorporated nine paired fields was conducted to quantify regional phenol

levels. In both the field and the regional survey studies, soil phenols were higher in CR than in FR fields. Together, higher phenol levels and lower N uptake$_{soil}$ in CR provide mechanistic evidence that the introduction of a season-long fallow to continuous rice systems enhances soil N availability by reducing organic substrate recalcitrance. Future work should identify the duration needed for soil phenols accumulation to impair soil N cycling under continuous rice cultivation, as well as any roles of soil microbial populations in these soil N cycling patterns.

Highlights

- N cycling in continuous rice and rice following a fallow were compared in a ^{15}N study
- Continuous rice had reduced soil N uptake in the ^{15}N study
- Fertilizer N uptakes for the two systems were similar
- Reduced soil N uptake in continuous rice was attributed to higher soil phenol levels

Keywords

^{15}N, N availability, N binding, soil organic matter, lignin-derived phenols, soil phenols

1. Introduction

Rice (*Oryza sativa* L.) is integral in meeting global food and nutritional demand (Mohidem et al., 2022). However, its production is becoming increasingly challenging due to climate change and its associated impacts such as temperature, precipitation, and salt intrusion (Espe et al., 2016; Nguyen et al., 2017; Yuan et al., 2021, 2022). California is a major producer of rice, accounting for 20% of total production in the U.S. (USDA, 2023). Specific to California, the increased frequency of winter droughts and spring rains leads to water restrictions and prevented planting respectively (Funk et al., 2014). During periods of erratic precipitation, growers typically fallow their fields or rotate to other more drought-tolerant upland crops (Rosenberg et al., 2022; Salvato et al., 2024). The increased incidences of a year-long fallow challenges the "norm" of yearly rice monoculture (i.e. continuous rice), and will have agronomic effects on productivity and nitrogen (N) cycling. Rice production in California is unique compared to most other rice-growing regions around the world. Typically, rice is grown as a monoculture in California as the heavy clay soils are not well suited for the cultivation of other crops (Hill et al., 2006; Rosenberg et al., 2022). Also, rice is grown in a direct water-seeded system where pregerminated rice seeds are broadcasted onto flooded fields. The flood is maintained on the field for the duration of the growing season until drainage for harvest. Elsewhere in the world, rice is often dry-seeded or transplanted. Finally, due to restrictions on rice straw burning in California, rice residue is incorporated into the soil following harvest, and the fields are flooded over the winter. The winter flood is the conventional method of residue management in the region as it promotes residue decomposition (Linquist et al., 2006).

In other regions, the introduction of aeration periods to continuous rice systems has been shown to increase yields and N availability. For example, in the Philippines, an average decline of -291 kg ha^{-1} yr^{-1}, was observed between 1968 and 1991 when rice was continuously tripled-cropped in a long-term experiment (Dobermann et al., 2000; Ladha et al., 2021). In the same experiment, the introduction of fallow periods after 1991 resulted in the reversal of the yield decline and improved agronomic performance (Dobermann et al., 2000). Similarly, in the U.S., a long-term experiment in Arkansas found that continuous rice systems averaged 19% less yield compared to rice in rotation with upland crops (Anders et al., 2004). The reduction of rice yields in continuous rice systems observed in these two regions was primarily attributed to soil N deficiency (Cassman et al., 1995; Olk et al., 2006). Therefore, in California, the introduction of aerobic conditions (via fallow or upland crop) to continuous rice fields is an area for investigation, as it has the potential to alter grain yield and N cycling.

Mechanistically, increased levels of N-stabilizing lignin-derived phenols in continuous rice systems had been empirically demonstrated to reduce soil N availability (Cassman et al., 1995; Olk et al., 1996). Soil phenols accumulate primarily due to crop residue incorporation and decomposition during prolonged flooding. Lignin is abundant in rice residues and its phenol derivatives have known N-binding effects (Min et al., 2015). The lignin origin of phenols and their effect on N binding were confirmed through solid-state nuclear magnetic resonance (NMR) (Schmidt-Rohr et al., 2004). Quantitative ^{13}C NMR combined with advanced spectral editing showed that more than 45% of all carbon (C) in a young humic fraction from a triple-cropped rice field originated from lignin derivatives, providing support that rice residue decomposition under anaerobic conditions was a significant source of phenol accumulation. Due to the aromatic structure of phenolics, N bonded to the benzene ring of the phenols will have a strong signal in the region of the NMR spectra (134 ppm) that corresponds to N-$C_{aromatic}$ bonds (covalent anilide bond). This covalent bond is highly stable, likely reducing N mineralization rates - at least under submerged conditions. This signal at 134 ppm was much higher in the young humic acid fraction from the triple-cropped continuous rice field than from a dryland rice field, confirming the strong N-binding effect of soil phenols in flooded rice fields.

In California, the conventional cropping system of continuous rice combined with winter flooding is a suitable environment for phenol accumulation. Despite having only one crop per year, the fields are flooded for roughly eight out of twelve months. Prolonged flooding, coupled with high residue retention, is ideal for increased levels and preservation of lignin-derived soil phenols. For example, in a 4-year experiment comparing continuous paddy rice to a rice-maize rotation, phenol levels in the continuous rice treatment steadily increased and became much higher than in the rice-maize rotation, suggesting that paddy rice monoculture favors higher phenol levels (Olk et al., 2009b). Given that soil phenols are generally considered a slow-cycling pool in soil organic matter (SOM) due to their recalcitrance (Min et al., 2015), higher phenol levels in continuous rice systems could result in a reduced rate of soil N mineralization for crop uptake.

The objective of this study was to quantify how the introduction of fallow impacts N cycling in Californian fields that had been in rice monoculture. Compared to previous studies quantifying the effect of short-term upland rotations or short-term fallows on N cycling in continuous rice systems, this is the first study to evaluate the effect of a season-long fallow in a unique system characterized by water-seeding and winter flooding. We tested the hypothesis that the introduction of a single-season fallow to a field that had been continuously cropped with rice will increase soil N availability. To address this hypothesis, we compared fallow rice to the conventional practice of continuous rice by using ^{15}N fertilizer to distinguish between crop N uptake from soil (N $uptake_{soil}$) and fertilizer (N $uptake_{fertilizer}$) pools. We investigated fertilizer N applications applied either preplant or as a topdress at panicle initiation (PI). Topdress N was used to test whether detected N deficiencies between treatments can be corrected with an additional N application at PI. Additionally, we investigated soil phenol levels as a potential mechanism for N cycling dynamics. The experiment was done in a rigorous replicated on-station study at the Rice Experiment Station (RES). We further conducted a region-wide soil survey study in the rice-growing region of the Sacramento Valley to confirm whether differences in soil phenol levels observed in the on-station experiment could be extrapolated over a broader region.

2. Materials and Methods

2.1 Site Description of Research Station Study

A field experiment was conducted at the California Rice Experiment Station (RES) near Biggs, Califonia (39°27’47” N, 121°43’35” W) in 2021 and 2022. The experiment was conducted in experimental plots (separated by levees) that were approximately 0.25 ha in area. The experimental plots had a history of rice cultivation for the past several decades. Soils at the experiment station are an Esquon-Neerdobe complex, classified as fine smectitic, thermic Xeric Epiaquerts, and Duraquerts, with a soil texture of approximately 290 g kg^{-1} sand, 260 g kg^{-1} silt, and 450 g kg^{-1} clay (Pittelkow et al., 2012). The climate at the site is Mediterranean. From May to October in the experimental years, the average daily temperatures were 22.1 °C in 2021 and 22.4 °C in 2022 (CIMIS, 2024).

Following land preparation and before fertilizer application, eight soil samples were collected from each plot at the plow layer (0-15 cm) and composited for background analyses (Midwest Laboratories, 2024). Soil pH values ranged between 5.3 -5.5 (1:1 soil/water), cation exchange capacity (CEC) (sodium acetate) between 28.5 – 33.1 $cmol_c$ kg^{-1} (Sumner & Miller, 1996), and organic matter content (Walkley-Black titration) between 23 – 35 g kg^{-1}.

2.2 Field Treatments

The experiment was set up as a randomized complete block design that was replicated three times with two treatments: continuous rice (CR) and fallow rice (FR). For the CR treatment, plots were identified where rice had been continuously grown for at least the previous six years. For the FR treatment, plots had been fallow the year before the experiment but had continuous rice cultivation for at least five years prior to the study year.

The availabilities of fertilizer and soil N applied preplant or as a topdress were evaluated using ^{15}N. The utilization of ^{15}N is an established method used to evaluate differences in N uptake as it enables direct measurement of N uptake from soil and fertilizer pools (Chen et al., 2016; Fan et al., 2007; Olk et al., 2009a). The form of ^{15}N fertilizer applied was ammonium sulfate with an enrichment level of 10.1%. To quantify crop N uptake during the growing season, five microplots were established within each plot for destructive plant sampling at designated times during the growing season. A microplot consisted of a polyvinyl chloride (PVC) ring 76.2 cm in diameter that was pushed into the soil to prevent the lateral movement of applied fertilizer. We examined the fate of fertilizer applied as preplant and as a topdress, following conventional rates applied by growers. Three of the microplots received N preplant at a rate of 150 kg N ha^{-1} and will be referred to as “preplant”. Preplant N microplot rings were sampled at three time points (one ring per sample at panicle initiation (PI), 50% heading, and maturity). Two of the microplots only received topdress N at a rate of 30 kg N ha^{-1} at PI and will be referred to as “topdress”. Topdress N microplot rings were sampled at two time points (one ring per sample at 50% heading and maturity). For preplant N, the ^{15}N fertilizer was hand-applied and incorporated into the soil after all land preparation was completed and before flooding commenced. For topdress, enriched ^{15}N fertilizer was hand-applied into the flood water at PI.

2.3 Field Management

In the CR treatment plots, following rice harvest in the year before the experiment, rice residue was chopped and incorporated into the soil. The plots were then flooded to aid decomposition, following conventional practice. In the FR treatment, the plots were fallowed in the year before the experiment but in the year prior to that, the straw was managed in the same way as the CR treatment following harvest. During the fallow period, the plots were disked in the spring and were subsequently unmanaged for the entire rice growing season until the start of rice cultivation in the next season. Before microplot establishment and planting, all experimental plots were chisel plowed, disced, leveled, and rolled. Potassium (2021: 39 kg K ha^{-1}, 2022: 33 kg K ha^{-1}) and phosphorous (2021: 20 kg P ha^{-1}, 2022: 22 kg P ha^{-1}) were applied to ensure that they were not limiting. Following flooding, pre-soaked rice seeds were broadcast seeded by hand. The field remained flooded for the duration of the season and was drained roughly three weeks after heading to prepare for harvest. Throughout the growing season, the crop was managed for weeds and pests following recommended standard practices.

2.4 Plant sampling, soil sampling, and laboratory analyses

Where ^{15}N fertilizer was applied preplant, all plants were sampled from one of the three microplots at PI, 50% heading, and physiological maturity. Where ^{15}N was applied as topdress, all plants were sampled from one of the two microplots at 50% heading and physiological maturity. For plant samples taken at PI and 50% heading, rice plants were hand-pulled from the microplots and were later cleaned and their roots removed. Samples were subsequently dried at 60°C before being ground into powder form for further analysis. Plant samples taken at physiological maturity were cut at soil level and oven-dried at 60°C. The grains and straw were separated with hand stripping to obtain their respective weights. Grain weights were then adjusted to 14% grain moisture and scaled to obtain yields (Mg ha^{-1}). The straw and grains were also ground to a powder. All plant samples were weighed on a microbalance into tin capsules before being analyzed for ^{15}N and total N uptake (N $uptake_{total}$) by combustion and EA-IRMS (elemental analyzer isotope ratio mass spectrometry) at the UC Davis Stable Isotope Facility (UC Davis SIF, 2023). For maturity samples, straw and grains were analyzed separately. N uptake (soil, fertilizer, total) for maturity was the sum of N present in both straw and grains. Soil samples within the microplots were also collected at maturity. Four subsamples were taken from each quadrant of the microplot using a Dutch auger (0-15 cm) to form a composite sample. The soil samples were air-dried and ground to a powder. The soil samples were then prepared for ^{15}N and total N analysis as described for the plant samples. Soil fertilizer N recovery, which is a direct measure of fertilizer ^{15}N left in the soil, was quantified from these samples.

Crop N uptake from soil (N $uptake_{soil}$) and fertilizer (N $uptake_{fertilizer}$) pools were then calculated based on established protocols (Cabrera & Kissel, 1989; Chen et al., 2016; Hauck &

Bremner, 1976). Background ^{15}N levels in our plots were determined to be at 0.3676% (^{15}N background (%)). Crop N uptake from fertilizer and soil pools was then calculated using equations (1) and (2), where 10.1% refers to the enrichment level of the ^{15}N labeled fertilizer and ^{15}N natural abundance (%) refers to the natural abundance of ^{15}N (0.3663%) (Blackshaw et al., 2002).

$$(1)\ N\ uptake_{fertilizer} = \frac{^{15}N\ detected(\%) - {}^{15}N\ background\ (\%)}{10.1\% - {}^{15}N\ natural\ abundance\ (\%)} \times N\ uptake_{total}\ (kg\ N\ ha^{-1})$$

$$(2)\ N\ uptake_{soil} = N\ uptake_{total}\ (kg\ N\ ha^{-1}) - N\ uptake_{fertilizer}\ (kg\ N\ ha^{-1})$$

For analysis of soil phenol levels in FR and CR treatments (representative of both preplant and topdress N), soils were collected from an area outside of the ^{15}N microplots at PI as differences in phenol levels can be detected at that timing (Olk et al., 2009a). Soil samples were taken from outside of the microplots to reduce disturbance to the growing crop and provide a representative sample for the plot. Eight subsamples were taken per plot using a Dutch auger (0-15 cm) to form a composite sample. The soil samples were air-dried at room temperature before being ground into powder form for subsequent analyses. Processed soil samples were analyzed following the established cupric oxide (CuO) oxidation protocol for soil phenol extraction (Olk et al., 2009a). Briefly, soil samples were placed into pressure bombs with NaOH and CuO as a catalytic oxidizing agent. The pressure bombs were then purged with Ar gas before being sealed and heated for 3 hours at 150 °C. Phenols were then extracted through centrifuging and ether washes. The solution was then filtered and derivatized through silylation with bis(trimethylsilyl)trifluoroacetamide. Finally, the processed solution was analyzed by gas chromatography for phenol concentrations. Per convention, phenol concentrations were reported by type as an enrichment level of soil organic carbon (g kg^{-1} OC) (Olk et al., 2009a).

2.5 Regional survey study

To investigate whether greater phenol levels in continuous rice versus rice following a fallow period or upland crop is a regional occurrence, a regional survey was conducted to complement the research station study. Phenol levels was the most suitable response to be evaluated as higher levels of soil phenols have been extensively documented to be indicative of N deficiencies in a rice system (Olk et al., 2006; Olk, et al., 2009a; Schmidt-Rohr et al., 2004). There were logistical challenges to getting other measurements as this was an on-farm study where fields are large and variable, and farmers applied different rates of fertilizer N. Thus, multi-site evaluation of yield or nitrogen use efficiency for treatment effects would be difficult to detect, and was more robustly addressed in the research station study.

On-farm sites, each consisting of one FR and one CR field, were sampled throughout the Sacramento Valley (Fig. 1). We sampled sites with diverse soil characteristics (Table 1), totaling four sites in 2021 and five sites in 2022. All CR fields were as described in section 2.2, where rice was continuously grown. FR fields from growers' fields were at times not fallow the previous season, but were a rotation with an upland crop such as safflower or tomato before returning to rice. Soil samples taken from growers' fields were sampled 60-70 days after seeding as differences in phenol levels can be detected at this sampling time and were similar to the timing in the research station study (Olk et al., 2009a). At each field, six soil samples were taken using a

Dutch auger (0-15 cm) to form a composite sample. The soils were processed and analyzed for standard soil characteristics and phenol levels as described above in sections 2.1 and 2.4.

Table 1

Background history and soil characteristics of sites used in the regional survey study including previous year management, pH, cation exchange capacity (CEC), and soil organic matter (SOM). Each site comprised one fallow rice (FR) field and one continuous rice (CR) field. Site numbers correspond to those presented in Figure 1.

Year	Site	Treatment	Previous year management	pH	CEC ($cmol_c\ kg^{-1}$)	SOM ($g\ kg^{-1}$)
2021	1	FR	Upland crop	6.6	47.7	37
		CR	Rice	6.2	46.5	47
	2	FR	Fallow	4.7	29.9	32
		CR	Rice	6.3	26.7	34
	3	FR	Upland crop	5.7	34.1	36
		CR	Rice	5.7	45.6	46
	4	FR	Fallow	5.4	41.2	41
		CR	Rice	6.5	32.7	37
2022	5	FR	Fallow	6.2	31.9	29
		CR	Rice	5.2	28.1	31
	6	FR	Fallow	5.9	24.1	29
		CR	Rice	6.4	19.5	32
	7	FR	Fallow	6.1	21.2	32
		CR	Rice	5.1	24.5	32
	8	FR	Fallow	5.7	33.2	41
		CR	Rice	5.8	41.8	43
	9	FR	Fallow	5.7	40.5	35
		CR	Rice	5.5	41.0	33

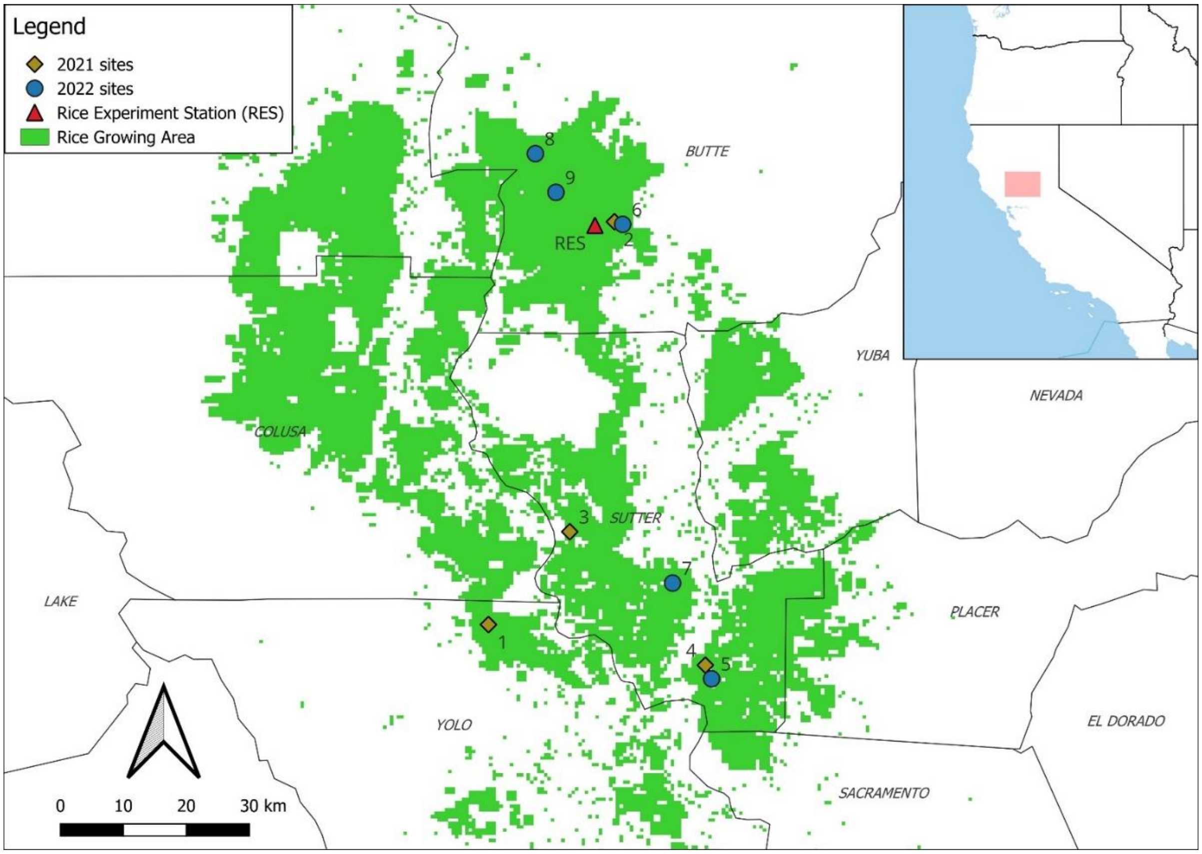

Figure 1. Locations of the nine on-farm sites for the regional survey study and the Rice Experiment Station (RES) in the Sacramento Valley rice-growing region of California, USA. Site numbers correspond to those presented in Table 1.

2.6 Data analysis

Statistical analysis of the data was performed in R-Studio (R Core Team, 2024). All data were analyzed using linear mixed effects models and the restricted likelihood method in the "lme4" package using the lmer function (Bates et al., 2015). In the research station study, there were five response variables, namely yield, N $uptake_{fertilizer}$, N $uptake_{soil}$, fertilizer N recovery (FNR), and phenol levels. For grain yield, FNR, and phenol levels, results were fitted with treatment, year, and their interaction as fixed effects and block as a random effect. For N

$uptake_{fertilizer}$ and N $uptake_{soil}$, results were fitted with treatment, year, sampling timing, and their interactions as fixed effects, and block and block:treatment as random effects to account for repeated measures. In the regional survey study, phenol levels were the only response variable. Results from the regional survey study were fitted with treatment and year as fixed effects and sampling site as a random effect. Analysis of variance (ANOVA) was conducted on the constructed models. Subsequently, individual comparisons for treatment effects were made. For all comparisons, P values were obtained using Tukey's HSD test and the "emmeans" package (Lenth, 2022). Results were presented by year only when there was both a significant year effect and treatment-by-year interaction. Consequently, only grain yield was presented by year. All other response variables were presented as pooled data from 2021 and 2022. Lastly, Pearson correlation was determined between various phenol types, crop N $uptake_{fertilizer}$, crop N $uptake_{soil}$, and soil fertilizer N recovery using the "ggpairs" function (Schloerke et al., 2021) (Fig. S1). Data visualizations were created with the "ggplot2" package (Wickham, 2016).

3. Results

3.1 Grain yield

Grain yields are presented by year for the ^{15}N study. Overall, grain yields were higher in 2021 compared to 2022. In both years, the FR treatment achieved higher yields compared to CR for preplant N (Fig. 2). CR had a lower grain yield (12.6 Mg ha^{-1}) than the FR treatment (14.9 Mg ha^{-1}) in 2021. Similarly, in 2022, CR yielded lower than FR at 10.9 Mg ha^{-1} and 12.6 Mg ha^{-1}, respectively. For topdress N, FR also yielded higher than CR, but this was only significant in 2022, where yields were 10.4 Mg ha^{-1} and 6.4 Mg ha^{-1} respectively. Generally, preplant N had higher yields due to the higher N rate compared to topdress N.

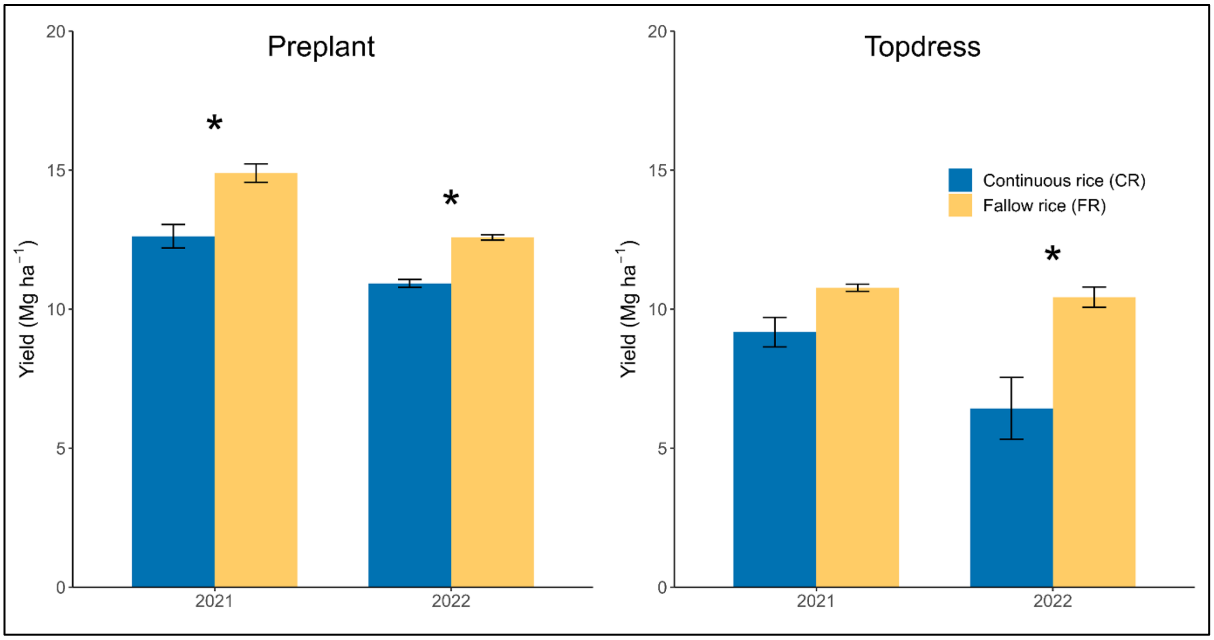

Figure 2. Grain yields (14% moisture) of the ^{15}N study. The left graph shows preplant applied N (150 kg N ha^{-1}) and the right graph shows topdress applied N (30 kg N ha^{-1} applied at panicle initiation). An asterisk indicates a significant difference (P<0.05; Tukey's HSD) between continuous rice (CR) and fallow rice (FR). Error bars represent the standard error.

3.2 Fertilizer N recovery

Fertilizer N recovery (FNR) from the crop and soil (0-15 cm) at maturity are presented as means between 2021 and 2022. No statistical differences in total FNR between CR (51.3%) and FR (56.4%) were observed for preplant N (Table 2). In contrast, total FNR was lower in CR (46.3%) compared to FR (58.8%) for topdress N, and this was driven by differences in crop FNR. At the end of the season, averaged between CR and FR and for the preplant and topdress applications respectively, an average of 22.9% and 18.4% of the applied fertilizer remained in the soil, and 31.0% and 34.2% of the applied fertilizer remained in the crop (Table 2).

Table 2

Crop, soil (0-15 cm), and total fertilizer nitrogen recoveries (FNR) calculated using ^{15}N values for continuous rice (CR) and fallow rice (FR) treatments. Values are expressed as a percentage (%) of the total applied ^{15}N fertilizer. In the Preplant (%) and Topdress (%) columns, FNR values (crop, soil, and total) followed by the same letter are not significantly different by Tukey's HSD test at P<0.05.

	Preplant (%)		Topdress (%)	
Recovery	CR	FR	CR	FR
Crop	30.1^{a}	31.8^{a}	29.3^{a}	39.1^{b}
Soil	21.2^{a}	24.6^{a}	17.0^{a}	19.7^{a}
Total	51.3^{a}	56.4^{a}	46.3^{a}	58.8^{b}

3.3 Preplant crop N uptake$_{soil}$ and N uptake$_{fertilizer}$

Preplant crop N uptake$_{soil}$ and N uptake$_{fertilizer}$ are presented as means between 2021 and 2022 as no year effect was detected. Using ^{15}N-labeled fertilizer, it is possible to distinguish between the source of crop N uptake as being from the soil or the fertilizer. When N was applied preplant, N uptake$_{total}$ was 138.5 kg N ha^{-1} for CR at maturity, where 93.4 kg N ha^{-1} came from N uptake$_{soil}$ and 45.1 kg N ha^{-1} came from N uptake$_{fertilizer}$. For FR, N uptake$_{total}$ was 157.9 kg N ha^{-1} at maturity, where 110.1 kg N ha^{-1} came from N uptake$_{soil}$ and 47.7 kg N ha^{-1} came from N uptake$_{fertilizer}$. The FR treatment had greater N uptake$_{soil}$ compared to CR (Fig. 3 - Preplant N uptake$_{soil}$) at all three sampling time points of PI, 50% heading, and maturity. The difference in N uptake$_{soil}$ between the treatments enlarged over the growing season from 12.9 kg N ha^{-1} (PI) to 15.1 kg N ha^{-1} (50% heading) to 16.7 kg N ha^{-1} (maturity). In contrast, N uptake$_{fertilizer}$ for CR and

FR were similar at 50% heading and maturity, with averages of 40.8 kg N ha^{-1} and 46.4 kg N ha^{-1} respectively (Fig. 3 – Preplant N $uptake_{fertilizer}$).

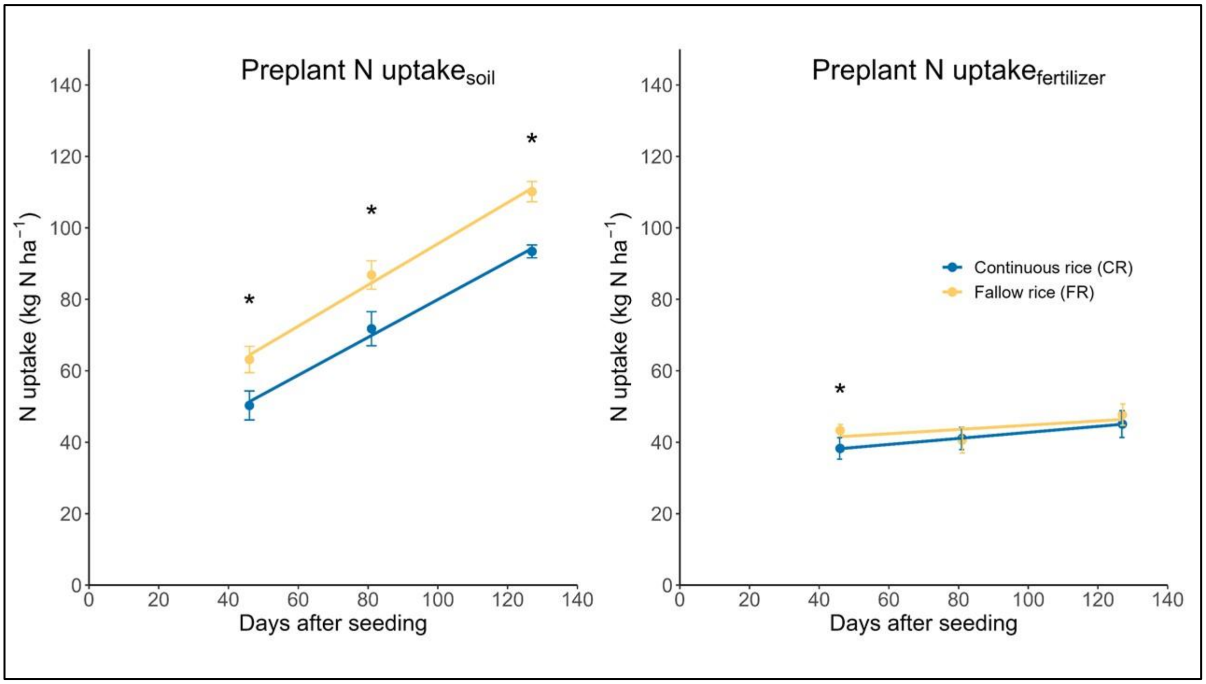

Figure 3. N $uptake_{soil}$ (left) and N $uptake_{fertilizer}$ (right) for preplant N (150 kg N ha^{-1}). Results were averaged across the 2021 and 2022 seasons. N uptake was quantified at panicle initiation (PI), 50% heading, and maturity in chronological order. Linear regression is shown to aid visualization. An asterisk indicates a significant difference (P<0.05; Tukey's HSD) between continuous rice (CR) and fallow rice (FR) for a given sampling point. Error bars represent the standard error.

3.4 Topdress crop N uptake$_{soil}$ and N uptake$_{fertilizer}$

Topdress crop N uptake$_{soil}$ and N uptake$_{fertilizer}$ are presented as means between 2021 and 2022. At maturity, N uptake$_{total}$ was 86.3 kg N ha^{-1} for CR, where 77.5 kg N ha^{-1} came from N uptake$_{soil}$ and 8.8 kg N ha^{-1} came from N uptake$_{fertilizer}$ when N was applied only as a topdress. For FR, N uptake$_{total}$ was 112.3 kg N ha^{-1} at maturity, where 100.6 kg N ha^{-1} came from N uptake$_{soil}$ and 11.7 kg N ha^{-1} came from N uptake$_{fertilizer}$. FR had both greater N uptake$_{soil}$ and N uptake$_{fertilizer}$ compared to CR (Fig 4). Compared to CR, FR had 22.7 kg N ha^{-1} more N uptake$_{soil}$ at 50% heading and 23.1 kg N ha^{-1} more N uptake$_{soil}$ uptake at maturity. N uptake$_{fertilizer}$ in the FR treatment was also greater, having 2.6 kg N ha^{-1} more N uptake$_{fertilizer}$ at 50% heading and 2.9 kg N ha^{-1} more N uptake$_{fertilizer}$ at maturity than the CR treatment.

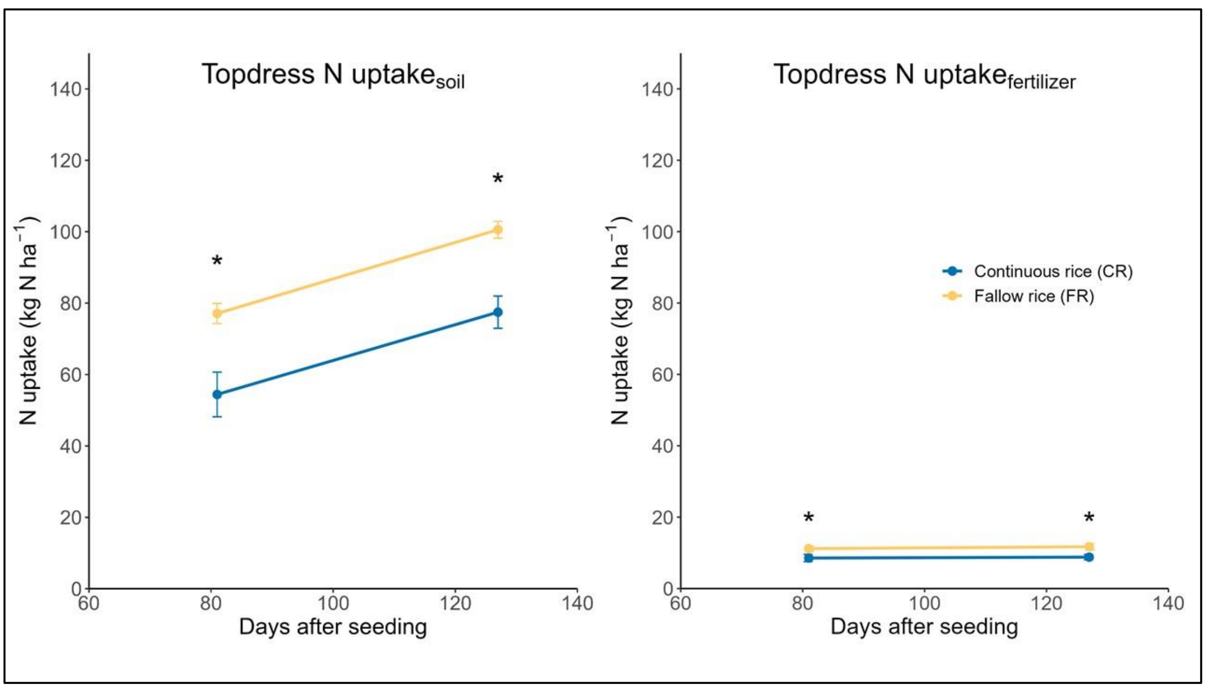

Figure 4. N uptake$_{soil}$ (left) and N uptake$_{fertilizer}$ (right) for topdress N (30 kg N ha^{-1} applied at panicle initiation). Results were averaged across the 2021 and 2022 seasons. N uptake was

quantified at 50% heading and maturity in chronological order. Linear regression is shown to aid visualization. An asterisk indicates a significant difference ($P<0.05$; Tukey's HSD) between continuous rice (CR) and fallow rice (FR) for a given sampling point. Error bars represent the standard error.

3.5 Soil phenol levels

Overall, the CR treatment had higher levels of soil phenols compared to FR (Fig. 5). At the RES (research station study), total phenol levels were greater in CR compared to FR, with phenol levels of 19.8 g kg^{-1} OC (organic carbon) and 17.2 g kg^{-1} OC respectively. The differences in total phenol levels were driven primarily by greater levels of cinnamic and vanillyl phenols in CR compared to FR. Based on a correlation analysis, the different types of phenols were all highly correlated with one another (Fig. S1). While total phenols and phenol types were not significantly correlated with N $uptake_{soil}$, there was a general trend showing that N $uptake_{soil}$ was lower at higher phenol levels.

In the regional survey study, the nine sites differed in their soil properties including organic matter, pH, and cation exchange capacity (Table 1). Nevertheless, higher levels of phenols were also observed in CR fields compared to FR fields, having values of 22.9 g kg^{-1} OC compared to 17.6 g kg^{-1} OC. By phenol type, CR also had higher phenol levels across all four types of phenols, namely cinnamic phenols, *p*-hydroxybenzoic phenols, syringyl phenols, and vanillyl phenols. The *p*-hydroxybenzoic acids are widely considered to be derived from multiple origins, including non-lignin compounds.

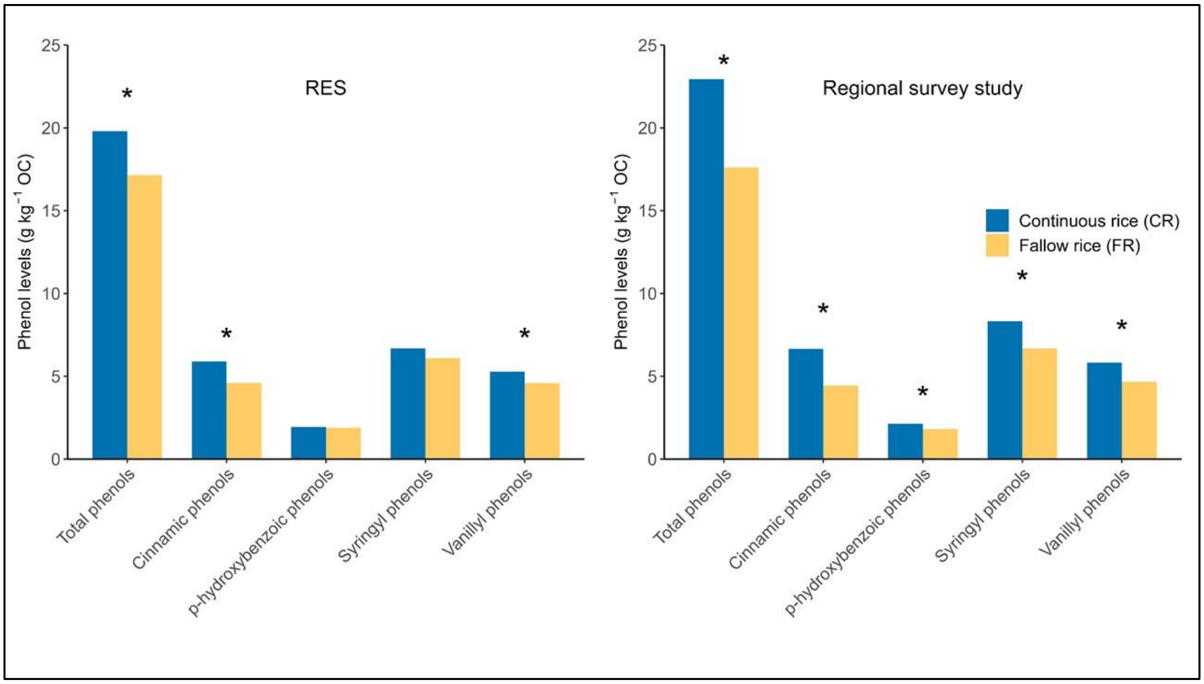

Figure 5. Phenol levels by type from the Rice Experiment Station (RES) and regional survey study. An asterisk indicates a significant difference between continuous rice (CR) and fallow rice (FR) ($P<0.05$; Tukey's HSD).

4 Discussion

4.1 Yield and fertilizer N recovery (FNR)

Rice yields in the two years of this study averaged 13.8 Mg ha^{-1} in 2021 and 11.8 Mg ha^{-1} in 2022 when preplant N was applied (Fig. 2). These differences between years reflect broader regional differences in rice yields across California, where yields in 2022 tended to be lower than 2021 (USDA, 2023). Yields were lower when only a topdress N application was applied, as this N rate was lower than the optimal and was applied too late in the season to achieve high yields (Cornelio & Linquist, 2023). However, our results show that topdress fertilizer N applied at PI could potentially provide an additional 8.8 kg N ha^{-1} N uptake$_{fertilizer}$ in CR. Total fertilizer ^{15}N recovery levels (FNR - ^{15}N recovered in crop and soil), in our study were consistent with other ^{15}N studies in rice, averaging 53.2% across years, treatments, and N applications (Bird et al., 2001; De Datta et al., 1988; Diekmann et al., 1993; Fan et al., 2007). We attribute yield differences between CR and FR treatments in each year to differences in N uptake, which are discussed below.

4.2 Reduction of N uptake in continuous rice

Our results are in agreement with, and expanded on, earlier related work, showing that the introduction of aerobic conditions to continuous rice systems (in this case a fallow year) increases N uptake$_{total}$. This difference between FR and CR is particularly notable in the N uptake$_{soil}$ fraction, as N uptake$_{fertilizer}$ was similar (Fig. 3 and 4). Similarly, in a ^{15}N study conducted in Arkansas, N uptake$_{soil}$ in continuous rice was 40 kg N ha^{-1} lower than in rice following an upland rotation (Olk et al., 2009a). In our study, CR had lower N uptake$_{soil}$ compared to FR and had the largest difference at maturity for both preplant (16.7 kg N ha^{-1}) and topdress (23.1 kg N ha^{-1})

applied N. Compared to previous studies that evaluated the effects of short-term fallows and upland rotations on continuous rice, the treatment of a season-long fallow is a unique aerobic soil environment and has not yet been evaluated in an empirical study. Across all ranges of aerobic exposure intensities - short-term fallows (approximately 2 months) in the Philippines (Dobermann et al., 2000), upland crop rotation in Arkansas (Olk et al., 2009a), and a season-long fallow in our study, continuous rice systems consistently had reduced soil N availability.

Reduced N uptake$_{soil}$ in continuous rice is thought to be due to higher levels of soil phenols (Olk et al., 2009a). Accordingly, we found that CR had higher levels of soil phenols compared to FR in both the rigorous replicated study and the regional survey study (Fig. 5). Increased levels of phenols can be attributed to the practice of CR management in California. Specifically, high levels of residue incorporation after harvest (the standard management strategy practiced by growers in the region) provide lignin substrates, and prolonged flooding (during both winter and summer) is an unfavorable environment for phenol degradation (Olk et al., 2006). Due to the anaerobic conditions during the growing season and winter flood of CR, microbial decomposition is slower than in aerobic environments and is mainly performed by fermentative bacteria and methanogenic archaea (Liesack et al., 2000). In contrast, the wet but aerobic winter experienced in FR was conducive to increased fungal activity, which is documented to be the most efficient microbial species at decomposing lignin and its derivatives (Floudas, 2021). Consequently, continuous rice soils were also associated with increased levels of cinnamic, syringyl, and vanillyl phenols, which are lignin residues that were derived from rice straw and root tissue (Olk et al., 2006; Olk et al., 2009a). Chemically, Schmidt-Rohr et al. (2004) verified that more than 45% of all carbon present in a humic acid fraction of a continuous rice soil were lignin derivatives.

While Schmidt-Rohr et al. (2004) suggested that increased anilide bonding between phenols and N accounted for reduced N availability in continuous rice during the growing season, this bonding might not be the only mechanism (Olk et al., 2009a). We hypothesize that increased N $uptake_{soil}$ in FR occurred due to the breakdown of phenols under aerobic conditions, releasing lignin-stabilized N into the soil pool for crop uptake (Olk et al., 2009a). The differences in phenols between the FR and CR treatments support this hypothesis. Looking more closely at the relationship between N availability and phenols, in further support of this hypothesis, we had expected negative correlations between the phenol levels (or phenol types) and N $uptake_{soil}$. Although correlations were negative, they were insignificant (Fig. S1). The lack of significance is likely due to the low number of observations, the relatively narrow range of both N $uptake_{soil}$ and phenol levels, and the intrinsic variability within field experiments (Fig. S1). Nonetheless, these data are in support of other studies performed under more controlled conditions that showed more significant negative correlations between phenol levels and N availability in a wide range of studies (Northup et al., 1995; Olk, 2008). Further research to investigate other potential mechanisms for phenol-N bonding to achieve a quantitative and stoichiometrically sound understanding of N stabilization by phenols would be valuable.

While we have suggested above that the accumulation of soil phenols was primarily responsible for reduced N uptake in the CR treatment, biological immobilization of N is another possible explanation. Studies have demonstrated the impact of high C:N residues on biological N immobilization and subsequent N uptake (Said-Pullicino et al., 2014). In the CR system, high levels of residue were incorporated into the soil after harvest in the previous fall. These fields were then flooded in the winter to aid residue decomposition (Linquist et al., 2006). By the start of the following growing season, undecomposed residue may contribute to fertilizer N immobilization. N $uptake_{fertilizer}$ in the CR system was 5.0 kg N ha^{-1} less than FR at PI for prelant N, possibly suggesting mild N immobilization (Fig. 3). At maturity, however, no differences in N $uptake_{fertilizer}$ were observed. Should CR have greater N immobilization than FR, an early short-term reduction in N $uptake_{soil}$ followed later by normal N $uptake_{soil}$ when C:N ratios are restored should be observed. Our study found that differences in N $uptake_{soil}$ increased as the growing season progressed. Such a trend would suggest that N immobilization may have had a small temporal effect early in season but the immobilized N became available later in the season. Consequently, reduced N $uptake_{total}$ in CR cannot be attributed mainly to biological N immobilization.

Reduced N uptake in CR also cannot be attributed to greater magnitudes of N losses through various pathways such as nitrate leaching, volatilization, and nitrous oxide losses (nitrification-denitrification). Firstly this was because similar levels of soil, crop, and total FNR were observed between the two systems in our study for preplant N. Secondly, the above-stated loss pathways are small in Californian rice systems and are sensitive to in-season management (Chuong et al., 2020; Liang et al., 2014; Linquist et al., 2018). Because CR and FR plots were identically managed during the growing season (especially water management), it was unlikely that either system had greater N losses that would result in reduced crop N uptake. Rather, our results suggest that differences in N uptake were likely due to increased soil organic matter recalcitrance in the CR system, especially increased phenol levels. This view is wholly consistent with the conclusions of Cassman et al. (1996) who found that total N in continuous rice soils was not indicative of crop N supply, due to perhaps multiple factors including increased recalcitrance of soil organic matter.

4.3 Management implications and future research

In California, the fertilizer N recommendation for rice growers is a preplant application at the full seasonal rate and then a crop assessment at PI to determine whether more N is needed (Rehmann et al., 2024). Importantly in this study, differences in N $uptake_{soil}$ between CR and FR were detectable at PI for preplant N (12.9 kg N ha^{-1}; Fig. 3), demonstrating that more soil N in FR was available early in the season. Previous work also demonstrated that the sufficiency of N uptake at PI is critical to secure high yields (Rehman et al., 2022), suggesting that a sufficient rate of fertilizer N should be applied preplant. Thus, given the differences in N $uptake_{soil}$, less preplant fertilizer N should be applied in FR systems. Based on average crop FNR values from this study of 32.3% (Table 2) and accounting for a difference of 16.7 kg N ha^{-1} in N $uptake_{soil}$ between CR and FR, 52 kg N ha^{-1} less fertilizer N could be applied in FR while still maintaining N uptake.

Our research has presented ^{15}N data providing direct measurements of N $uptake_{soil}$ and N $uptake_{fertilizer}$, and phenol levels as an explanatory factor for reduced N uptake in CR. While phenol levels were consistently higher in CR, this was likely not the only factor affecting N availability. Future research should investigate potential biological factors of this phenomenon, specifically differences in the structure and function of the microbial communities involved in the lignin degradation process during both the growing season and winter off-season. Additionally, there is a need to understand the number of crop cycles during which continuous rice needs to be practiced for N availability to decrease noticeably, and whether adjustments to conventional management of continuous rice can improve N cycling. In California, the practices of winter flooding and residue incorporation post-harvest are conventional management practices. Reducing flooding duration in combination with residue removal could be a direction for future research as they address sources of phenol preservation and lignin enrichment respectively. Specifically for the winter, longer durations of aerobic exposure can assist in the breakdown of phenols and residue (Gao et al., 2016). Short-duration, in-season drainages (e.g. mid-season drain), could also be evaluated as technical options. Strategies that increase aerobic soil conditions should also evaluate changes in soil organic matter quality over time to ensure it does not reduce productivity in the long term by depleting the organic N pool (Livsey et al., 2019; Zhang et al., 2024). Overall, results from the study provide strong evidence that FR has increased N $uptake_{soil,}$ and more field research can continue to evaluate optimal N management strategies in the context of increased fallow frequencies.

5 Conclusion

CR compared to FR had lower N $uptake_{total}$ than did FR, driven strongly by reduced N $uptake_{soil}$ in CR as shown in the ^{15}N study. The reduction of soil N availability for CR supports previous findings that reduced N availability in this system is related to an increase in soil phenols. In both the research station and the regional survey study, we found significantly higher levels of soil phenols in CR compared to FR systems. The increased phenol levels in the CR treatment suggest a greater degree of N binding with soil phenols, thereby reducing soil N availability and N $uptake_{soil}$. For future research, there are at least two broad directions. First, to enable optimal N fertilizer management, it would be crucial to establish the duration of rice monocropping before a noticeable change in N dynamics is observed. Second, research into the mechanisms of phenol-N stabilization and N mineralization in relationship to microbial structure and function will provide an improved understanding of N cycling differences between the CR and FR systems. With increased uncertainties in precipitation patterns, more fallowing will occur in California. By accounting for increased soil N availability in FR, growers have an initial framework to manage fallow rice by strategically reducing fertilizer N rates.

Data availability statement

Scripts and data supporting the findings of this document are available at:

https://github.com/XiaoZhangZhangRice/15N-and-Phenols-California

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Conflict of Interest

All authors declare no conflicts of interest.

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Chapter Three

Title

Greenhouse gas emissions altered by the introduction of a year-long fallow to continuous rice systems

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*Figures and tables are located at the end of Chapter Three.

Core Ideas

1. The effect of introducing a year-long fallow to continuous rice systems on GHG emissions was investigated.

2. Over three years, annual emissions were quantified in continuous rice (CR), fallow rice (FR), and fallow (F).

3. During the summer season, cumulative CH_4 emissions were 45-53% lower in FR than CR in two out of three years.

4. Summer CH_4 emissions in FR and CR contributed to more than 95% of their annual global warming potentials.

5. F had the lowest annual global warming potential, with winter N_2O emissions being the main contributor.

Plain language summary

California rice systems, characterized by continuous mono-cropping, are experiencing increased year-long fallow frequencies due to erratic precipitation. These fallows can alter greenhouse gas (GHG) emissions, especially methane (CH_4), in both the fallow and subsequent rice phases. Summer and winter GHG emissions of three systems - continuous rice (CR), fallow rice (FR), and fallow (F) - were quantified. CR had the highest CH_4 emissions in summer, averaging 476 kg CH_4 ha^{-1}. FR had 33% lower summer CH_4 emissions, averaging 318 kg CH_4 ha^{-1}. High summer CH_4 emissions contributed more than 95% to the annual global warming potential in CR and FR. F had the lowest annual global warming potential, with no summer CH_4 emissions and winter nitrous oxide (N_2O) emissions being the main contributor. Our results present complete annual CH_4 and N_2O emissions for the three systems, quantifying changes to GHG emissions when fallow is introduced to rice monoculture in California.

Keywords

Methane, nitrous oxide, particulate organic carbon, mineral-associated organic carbon, global warming potential

Abbreviations

CR – continuous rice

F – fallow

FR – fallow rice

GHG – greenhouse gas

GWP – global warming potential

MAOC – mineral-associated organic carbon

POC – particulate organic carbon

Abstract

Rice (*Oryza sativa* L.) production in California follows a norm of mono-cropping with little to no rotations or fallows. Both winter droughts, which lead to water restrictions, and spring rains, which inhibit field machinery operations, have resulted in increased fallow frequencies (no crop grown during the summer growing season). A three-year field study was conducted to investigate changes in GHG emissions (CH_4 and N_2O) following the introduction of a year-long fallow to continuous rice systems in both the fallow and rice-following-fallow phases with three treatments - continuous rice (CR), fallow rice (FR), and fallow (F) - during summer and winter. In the summer, FR had a 45-53% reduction in cumulative CH_4 emissions compared to FR in two of three years (no reduction in one year likely due to differences in residue management). F had no CH_4 emissions in the summer. Summer N_2O emissions were low for all three treatments. Summer global warming potential (GWP) accounted for more than 96% of annual GWP in CR (13,937 kg CO_2 eq ha^{-1}) and FR (9,236 kg CO_2 eq ha^{-1}). For F, the winter season accounted for 94% of the annual GWP (413 kg CO_2 eq ha^{-1}) due to N_2O emissions. Additionally, we showed that particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) levels in CR and FR were similar in the year with no treatment effect on CH_4 emissions. Overall, this study took an important step in quantifying changes to GHG emissions when fallow is introduced to rice monoculture in California.

1. Introduction

Erratic precipitation patterns in California have disrupted the norm of continuous rice production in the Sacramento Valley, resulting in increased fallowing of rice land throughout the growing region. First, California has had a higher frequency of drought events in the past decade, driven mainly by reduced winter precipitation (Funk et al., 2014; Gebremichael et al., 2021). Drought directly reduces surface water availability, causing water restrictions from irrigation districts and resulting in farmers fallowing their land (Dale et al., 2013; Keppen & Dutcher, 2015). Second, there are often unpredicted, and high precipitation events during land preparation in spring (April to May). Machinery used for land preparation operations such as chisel plows and leveling equipment cannot enter wet fields, delaying planting. Should planting not be completed by a designated date due to weather conditions, growers have the option to claim crop insurance through the “Prevented Planting” scheme, providing a financial safety net (USDA-RMA, 2024). Together, reduced precipitation during winter and increased precipitation during land preparation, directly and indirectly, contribute to increased fallow acreages in a cropping system previously characterized by continuous rice monoculture in the past decade (Rosenberg et al., 2022; Salvato et al., 2024). Rice is a major contributor to agricultural greenhouse gas (GHG) emissions (Carlson et al., 2017; Zhang et al., 2024), especially methane (CH_4), due to flooded soils and high carbon (C) inputs through rice straw. To understand how much the introduction of fallow periods may affect CH_4 emissions, and to a smaller extent, nitrous oxide (N_2O) emissions, it is pertinent to evaluate the effect of a year-long fallow on GHG emissions at a system level.

In California, the rice system typically includes a summer growing season and a winter off season. After harvest, rice residue is usually incorporated into the soil, and fields are flooded in the winter to aid decomposition (Linquist et al., 2006). For a fallow season, fields are tilled in the spring following the previous rice harvest, and subsequently unmanaged through the summer and winter. Consequently, fallow fields are not flooded in the summer and the winter, resulting in higher redox potentials and potentially leading to little to no CH_4 emissions (Minamikawa & Sakai, 2005). While a large number of studies have evaluated GHG emissions during the rice growing season, a smaller number have documented winter emissions following rice in California (Adviento-Borbe et al., 2013; Pittelkow et al., 2014). Wet winters during the fallow may also be conducive for soil organic nitrogen (N) mineralization and aerobic decomposition, leading to N_2O emissions during nitrification or denitrification (Gaihre et al., 2020). Aerobic decomposition can also reduce soil C pools, leading to reduced CH_4 emissions in the rice season following the year-long fallow.

While no studies have measured GHG emissions following a full year of fallow or during a year-long fallow, several studies have quantified GHG emissions in rice following a rotation with an upland crop. Upland rotations have similarities with a fallow in that the soil is aerobic for a full year prior to the rice phase. Pertaining specifically to the summer rice growing season, the rice crop following upland crop rotations had reduced CH_4 emissions (Eusufzai et al., 2010; Akiyama & Yagi, 2011; Weller et al., 2016). For example, rotations from aerobic rice to flooded rice or maize to rice had CH_4 reductions between 54-60% compared to a rice-rice mono-crop system (Weller et al., 2016). Nishimura et al. (2011) also reported similar findings, with reductions of 13–92% and 54–87% in CH_4 emissions from the rice following aerobic rice or soybean-wheat rotation, respectively. Reduced CH_4 emissions from rice following a rotation were attributed to reduced residue input compared to paddy rice, resulting in reduced soil C substrate availability for CH_4

production (Akiyama & Yagi, 2011). Other studies have also suggested that aeration can increase soil Fe (III) levels (Conrad, 2002; van Bodegom et al., 2000), or lower the abundance of methanogenic archaea (Breidenbach et al., 2016; Liu et al., 2015), thereby lowering CH_4 emissions in the rice crop following rotation. In all of the studies, nitrous oxide (N_2O) emissions were small, and CH_4 was the main contributor to global warming potential (GWP, kg CO_2 eq ha^{-1}) during rice phases.

Prolonged soil aerations such as upland rotations may alter soil C quantity and quality, resulting in reduced CH_4 emissions. Generally, aerobic soils have a higher C mineralization potential compared to anaerobic soils (Gale & Gilmour, 1988), and rice soils following aeration events may lose soil C. For example, Sun et al. (2019) found that the upland rotations of double maize and rice-maize had more soil C loss compared to traditional double rice cropping. Similarly, Pampolino et al. (2006) reported that changing the rice-rice system to a rice-maize rotation caused a 14% reduction in soil organic carbon (SOC). Given the similarities between upland rotation and fallow, there is potential for soil C quantity and quality to change in the fallow-rice system we are examining. Moreover, changes to fast and slow-cycling soil C pools of particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) (Yu et al., 2022), can drive CH_4 emissions. In oceanic and river systems, CH_4 emissions were positively correlated with particulate organic matter levels (Berberich et al., 2020; Karl & Tilbrook, 1994). Specifically, the year-long fallow may reduce soil C level, especially POC, thereby reducing CH_4 emissions. While POC and MAOC have been previously evaluated for nitrogen, copper, and zinc cycling in rice systems (Bu et al., 2024; Shi et al., 2018), they have not been evaluated as potential indicators of CH_4 emissions in rice systems.

Changes to CH_4 emissions with a year-long fallow will likely be due to a combination of reduced residue input, extended aeration promoting microbial respiration, and changes to soil C fractions. Utilizing a three-year field study, our main objective was to quantify CH_4 and N_2O emissions from three treatments: fallow rice (FR; rice following a fallow), continuous rice (CR), and fallow (F) during the summer and winter seasons (Figure 1). We hypothesized that during summer, fallow rice (FR) will have reduced cumulative seasonal CH_4 emissions compared to continuous rice (CR), but N_2O emissions will be similar. Additionally, we hypothesized that fallow (F) will have negligible amounts of CH_4 and N_2O emissions during the summer because the soil is dry. During winter, we hypothesize that CH_4 emissions will be similar between the flooded CR and FR, but negligible for the unflooded F. Due to low temperatures in the winter, we expect N_2O emissions to be low and similar between the three treatments. Lastly, we also investigated POC and MAOC levels in CR and FR soils sampled before each growing season as an indicator of CH_4 emission levels. Compared to previous studies that examined the effect of upland crop rotations on GHG emissions, this was the first empirical study to document the effect of a year-long fallow on GHG emissions and soil C change. Overall, this study plays a pivotal role in quantifying GHG emissions in continuous rice, fallow rice, and fallow phases, for an agroecosystem experiencing a clear increase in fallow frequency.

2. Materials and methods

2.1 Study site description

The study was conducted at the Rice Experiment Station in Biggs, California (39°27’47” N, 121°43’35” W) from 2021 to 2023 (Figure 1). Research plots had a history of rice cultivation for the past few decades. The plots used for this study were separated by levees and were approximately 0.25 ha in size. Soils at the site are an Esquon-Neerdobe Complex, classified as fine, smectitic, thermic Xeric Epiaquerts and Duraquerts, with a soil texture of 290 g kg^{-1} sand, 260 g kg^{-1} silt, and 450 g kg^{-1} clay (Pittelkow et al., 2012). Selected soil characteristics (0-15 cm) for the plots include: pH between 5.3 - 5.7 (1:1 soil/water), cation exchange capacity (CEC) (sodium acetate) between 26.3 – 33.4 cmolc kg^{-1}, and organic matter content (Walkley-Black titration) between 23 – 39 g kg^{-1} (Midwest Laboratories, 2024). The climate at the site is Mediterranean, with little to no precipitation in the summer and most precipitation occurring in the winter between November to March (Figure 2). During the experimental years, the average daily temperatures were 16.8 °C in 2021, 16.6 °C in 2022, and 16.0 °C in 2023 (CIMIS, 2024). During the winter season (November to April), total precipitation was 148 mm in 2021/22, and 500 mm in 2022/23 (CIMIS, 2024).

2.2 Treatments and experimental design

There were three main treatments, continuous rice (CR), fallow rice (FR), and fallow (F) (Figure 1) (Zhang, 2024). The CR treatment is rice following six years of annual rice cropping, while FR is rice following one year of fallow (prior to the one-year fallow, rice had been continuously grown for five years). The F treatment consisted of fallow plots following five years of continuous rice. The experiment was set up as a completely randomized design with three replicates during each emission period, where each plot received a treatment of CR, FR, or F. Emission periods for each treatment were separated into summer (CR_{summer}, FR_{summer}, F_{summer}) and winter (CR_{winter}, FR_{winter}, F_{winter}) seasons (Figure 1). Summer emission periods generally lasted from May to October while winter emission periods lasted from November to May. The rice summer season (CR_{summer} and FR_{summer}) had emissions quantified during 2021, 2022, and 2023. The F_{summer} was quantified in 2021 and 2022. Winter emissions (CR_{winter}, F_{winter}, and FR_{winter}) were quantified in 2021/22 and 2022/23. Total annual emissions (CR_{annual}, FR_{annual}, and F_{annual}) were the sum of GHG emissions from the summer and winter emission periods for a given treatment.

2.3 Plot management

CR and FR were treated identically during the rice phase. When plots were sufficiently dry in the spring, they were chisel plowed, disced, leveled, and rolled before the growing season (Figure 1). Potassium (2021: 39 kg K ha^{-1}, 2022: 33 kg K ha^{-1}, 2023: 40 kg K ha^{-1}) and phosphorous (2021: 20 kg P ha^{-1}, 2022 and 2023: 22 kg P ha^{-1}) were applied to ensure that they were non-limiting. Nitrogen (N) was applied as aqueous ammonia between leveling and rolling. In 2021 and 2022, 225 kg N ha^{-1} was applied to plots, while in 2023, an N rate of 180 kg N ha^{-1} was used. M206 variety rice seeds were soaked in water for at least 24 hrs before being hand applied to flooded plots. Plots were continuously flooded for the entire growing season, following standard practice. Approximately three weeks before harvest, plots were drained. Pests and weeds were controlled as per standard practice. Rice was harvested when moisture content was lower than 26%. In 2021 and 2022, rice plants were harvested by hand from 1 m^2 harvest rings that were placed in the plots before flooding. Plants were cut at the root level before being dried at 60°C. Grains were separated from straw using hand-stripping for yield quantification. In 2023, plants were harvested using an ALMACO small plot combine. Yields were reported at 14% moisture.

After harvest, residue was incorporated and the plots were flooded in the winter to aid residue decomposition (Figure 1, FR_{winter} and CR_{winter}) (Linquist et al., 2006). In 2020 and 2022, residue was incorporated using a disc (Figure 3) and followed by flooding. However, in 2021, due to high precipitation in October before discing operations (CIMIS, 2024), rice residue was incorporated through “stomping” (Figure 3). Stomping involves pressing residue into flooded soils with a large cage roller to encourage good residue-soil-water contact (Blank et al., 1993).

For plots entering the fallow treatment from continuous rice, the plots were managed post-harvest and through the winter as described above. At the beginning of F_{summer}, plots were disced in the spring and were subsequently unmanaged during the summer and winter.

2.4 Greenhouse gas measurements

For all emissions periods, gas samples were collected with the static vented chamber technique (Adviento-Borbe et al., 2013). Each cylindrical chamber consisted of a permanent base, a variable height extension to accommodate rice growth, and a chamber lid with a vent for pressure equalization. Across all emission periods, when soil moisture was expected to change rapidly (i.e. initial flooding, drainage, and rains when soils were not saturated), gases were sampled every other day or every day as emission fluxes were expected to change rapidly.

Prior to the growing season (FR_{summer} and CR_{summer}), two bases were permanently installed in each plot. To reduce disturbance to the rice, gas sampling alternated between the two chambers and sampling took place at a one-week interval (Figure 1) (Adviento-Borbe et al., 2013;

Perry et al., 2022, 2024). During the winter emission periods (FR_{winter}, F_{winter}, CR_{winter}), gases were sampled at two-week intervals. Lastly, a three-week sampling schedule was used for F_{summer}. A longer sampling interval was used as little to no CH_4 and N_2O fluxes were expected in F_{summer} with consistently dry soils (no summer rain and irrigation) (Ewing et al., 2007; Wu et al., 2010).

Gas sampling took place between 0900 and 1300 PST as gas fluxes and soil temperatures were representative of daily average values (Adviento-Borbe et al., 2013). At each gas sampling event, 25 mL of gas sample was taken from each chamber at times 0, 21, 42, and 63 minutes after chamber seal and injected into pre-evacuated 12.5 mL glass vials (Labco Ltd., Buckinghamshire, UK). To ensure homogeneity before taking a gas sample from the enclosed chamber, a fan was used to mix gases.

Gas samples were analyzed for CH_4 and N_2O peak areas on a Shimadzu GC-2014 gas chromatograph (Shimadzu Scientific Instruments, Columbia, MD, USA). The gas chromatograph had a detection limit of 1.83×10^{-4} mg L^{-1} for both CH_4 and N_2O. CH_4 and N_2O were separated isothermally at 75°C by a stainless steel column packed with Hayesep D, 80/100 mesh. CH_4 concentration was measured using the FID (flame ion detector) set at 250 °C, while N_2O was measured using a ^{63}Ni ECD (Electron Capture detector) set at 325 °C. During each chromatography analysis cycle, a calibration curve was constructed using gas standards with known concentrations (N_2O: 0, 0.3, 1, and 9.95 ppm, CH_4: 0, 1.8, 10.18, and 503 ppm) and their associated peak areas. The calibration was accepted and used to determine the sample gas concentrations if it followed a linear relationship and r^2 was greater than 0.99. Fluxes were then calculated with linear increases in gas concentrations, chamber volumes measured at each sampling event, and the Ideal Gas Law using equation (1):

$$(1)\ F = \frac{\Delta C}{\Delta t} \times \frac{V}{A}$$

where F is the daily flux (g CH_4 ha^{-1} day^{-1}; g N_2O ha^{-1} day^{-1}), $\Delta C/\Delta t$ is the increase of gas concentration in the chamber (g L^{-1} day^{-1}), V is chamber volume (L), and A is chamber area (ha).

Per convention with other GHG studies in rice, daily flux values were accepted if two criteria were met. The two criteria are: (1) a linear increase in gas concentrations over time was detected for each observation (r^2>0.845), and (2) the change in gas concentrations was above the gas chromatography instrument's minimum detection limit (>1.83 × 10^{-4} mg L^{-1}) (Adviento-Borbe et al., 2013; Perry et al., 2022, 2024). Gas fluxes without linear increases were considered missing data and changes in gas concentrations below the detection limit were set to zero for data analysis. Linear interpolation was conducted between sampling events to determine daily flux values. Cumulative seasonal emissions (kg CH_4 ha^{-1} or kg N_2O ha^{-1}) were determined as the sum of all daily emissions. Global warming potential (GWP, kg CO_2 eq ha^{-1}) was calculated for a 100-year time horizon using radiative forcing potentials relative to CO_2, corresponding to 28 for CH_4, and 265 for N_2O respectively (Pachauri & Meyer, 2014).

2.5 Quantification of particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) fractions

To quantify changes in soil C, we examined particulate organic (POC) and mineral-associated organic carbon (MAOC) fractions, which are generally considered fast and slow cycling fractions, respectively (Yu et al., 2022). Soil samples were collected after land preparation and before fertilizer application before the rice phases (CR_{summer} and FR_{summer}). Eight soil samples were collected per plot from the plow layer (0-15 cm) and composited for analysis. Soils were separated into POC and MAOC fractions through aggregate dispersion, wet serving, and particle size physical fractionation following protocols outlined in Brewer et al. (2023) and Six et al. (1998). Briefly, 10 g of air-dried soil was shaken for 18 hours on a rotary shaker with 30 ml of 5% (w/v) sodium hexametaphosphate ($Na_6(PO_3)_6$) and five glass beads for sufficient dispersion. Dispersed soils were wet filtered through a 53 µm sieve using deionized water and a vibratory sieve shaker (Fritsch Analysette 3 Pro – Idar-Oberstein, Germany). The residue (>53 µm) was considered POC while the filtrate (<53 µm) was considered MAOC. Both fractions were dried at 60 °C until constant mass before being ground to a powder for total C analysis on an elemental analyzer (Costech ESC 4010 Elemental Analyzer – Valencia, CA, USA). Additionally, we also quantified soil organic carbon (SOC), which is the sum of POC and MAOC fractions. POC, MAOC, and SOC concentrations were then calculated using equations (2), (3), and (4):

$$(2)\ POC\ (g\ C\ kg^{-1}\ soil) = \frac{M_{POC}\ (g)}{M_{Total}\ (g)} \times P_{POC}(g\ C\ 100g^{\ -1}\ soil) \times 10$$

$$(3)\ MAOC\ (g\ C\ kg^{-1}\ soil) = \frac{M_{MAOC}\ (g)}{M_{Total}\ (g)} \times P_{MAOC}(g\ C\ 100g^{\ -1}\ soil) \times 10$$

$$(4)\ SOC\ (g\ C\ kg^{-1}\ soil) = POC\ (g\ C\ kg^{-1}\ soil) +\ MAOC\ (g\ C\ kg^{-1}\ soil)$$

where POC, MAOC, and SOC are expressed in g C kg^{-1} soil, M_{MAOC} and M_{POC} refer to the mass of the associated fraction, M_{Total} refers to the total mass of the soil sample, P_{POC} and P_{MAOC} refer to the mass percentage of C in the associated fraction, and 10 is a scaling factor to convert g C 100 g^{-1} to g C kg^{-1}.

2.6 Statistical analyses and data visualization

Climatic data were collected from the California Irrigation Management Information System (CIMIS). Monthly precipitation (mm) and average monthly temperature (°C) were retrieved from the Biggs station (Figure 2).

Statistical analyses were performed in R-Studio using linear models fitted with the lm function (R Core Team, 2024). The data and scripts that support the findings of this study are available in an online repository (https://github.com/XiaoZhangZhangRice/FallowRice_ContinuousRice_GHG_Carbon). Data processing was primarily performed using the "dplyr" package (Wickham et al., 2023). All response variables, namely cumulative seasonal CH_4 and N_2O emissions, GWP, yield, SOC, POC, and MAOC, were fitted with treatment and year as fixed effects. Analysis of variance (ANOVA) was then conducted. Due to strong treatment-by-year interactions being present for cumulative CH_4 emission and GWP in the summer, results were presented by year and emission period. As such, for all response variables, individual pairwise comparisons were made for each year and its associated emission period. P-values were obtained using Tukey's multiple pairwise comparisons and mean values were obtained using the "emmeans" package (Lenth, 2022). Because F_{summer} data were not collected in 2023, the dataset was unbalanced. To test if average emissions between the three treatments were different in the summer, additional linear models were fitted with only treatment as a fixed effect and analyzed in a similar way as described above. Data visualizations were created with the "ggplot2" and "ggpubr" packages (Kassambara, 2023; Wickham, 2016).

3. Results

3.1 Climate and yield

There were distinct precipitation differences between the winter periods (November – March) in this study (Figure 2). Winter 2022/23 had the highest precipitation, totaling 492 mm. In contrast, the winters of 2020/21 and 2021/22 had 148 mm and 139 mm of precipitation respectively. During the summer, there were two unusual precipitation events. First, there was a large storm event in October 2021, totaling 117 mm of precipitation. High rainfall persisted through November and December 2021. The wet fields prevented dry tillage (discing) and thus rice residue was incorporated using stomping instead (see section 2.3 Plot management). Second, there was a storm event in September 2022, totaling 51 mm, prompting high-frequency gas sampling events (Figures 4 and 5) and causing crop lodging. Monthly temperature patterns remained generally consistent across years during the winter and summer seasons. Grain yields were similar in 2021 and 2022 (hand harvest), but CR had higher yields in 2023 (combine harvest) (Table 1). Across treatments and years, yield averaged 11,272 kg ha^{-1}.

3.2 Summer greenhouse gas emissions

During the summer growing season, cumulative CH_4 emissions differed between treatments (Table 1). F_{summer} had the lowest emissions, with no CH_4 emissions in 2021 and 2022. Lower cumulative CH_4 emissions were observed in FR_{summer} compared to CR_{summer} in two out of three years - 2021 and 2023. In 2021, the reduction was 45% ($P<0.05$), and in 2023, the reduction was 53% ($P<0.05$). In 2022, emissions were similar between CR_{summer} and FR_{summer}. On average,

CR_{summer} had the highest cumulative emission at 476 kg CH_4 ha^{-1}. FR_{summer} had lower cumulative CH_4 emissions at 318 kg CH_4 ha^{-1}, representing a 33% reduction on average ($P<0.05$).

In 2021 and 2023, where CR_{summer} had greater cumulative CH_4 emissions than FR_{summer}, daily CH_4 flux was consistently greater in CR_{summer} compared to FR_{summer} (Figure 4). Peak daily fluxes were 7731 and 8207 g CH_4 ha^{-1} day^{-1} for CR, and 4628 and 4416 g CH_4 ha^{-1} day^{-1} for FR, in 2021 and 2023 respectively. In 2022, daily CH_4 fluxes for CR_{summer} and FR_{summer} were generally similar. For all three years, there were distinct end-of-season spikes in daily CH_4 flux in September after drainage (Figure 4).

Cumulative summer N_2O emissions were similar for the three treatments and no statistical differences were detected within or across years (Table 1). On average, cumulative N_2O emissions were 0.25 kg N_2O ha^{-1} for CR_{summer}, 0.15 kg N_2O ha^{-1} for FR_{summer}, and 0.09 kg N_2O ha^{-1} for F_{summer}. Daily N_2O fluxes were only detected during May and September (Figure 5). These periods corresponded to initial flooding (May) and final floodwater drainage (September) for CR_{summer} and FR_{summer}. During the continuous flood, no N_2O emissions were detected. In 2022, precipitation events during September resulted in daily N_2O fluxes from F_{summer} plots.

Cumulative N_2O emissions were responsible for all global warming potential (GWP) in F_{summer} as there were no CH_4 emissions. On average, F_{summer} had a GWP of 23 kg CO_2 eq ha^{-1}. For CR_{summer} and FR_{summer}, cumulative N_2O emissions contributed to less than 1% of GWP on average and GWP was driven almost exclusively by CH_4 emissions. Averaged across years, FR_{summer} had a GWP of 8,941 kg CO_2 eq ha^{-1} while CR_{summer} had a GWP of 13,396 kg CO_2 eq ha^{-1}, representing a 33% reduction with FR_{summer} ($P<0.05$).

3.3 Winter greenhouse gas emissions

Cumulative CH_4 emissions were similar between CR_{winter} and FR_{winter} in 2021/22 and 2022/23, averaging 17 and 7.1 kg CH_4 ha^{-1} (Table 1). The majority of CH_4 emissions for CR_{winter} and FR_{winter} occurred from January to March in both 2021/22 and 2022/23 (Figure 4). These were periods when plots were flooded for a long duration or after drainage. F_{winter} interestingly had negative cumulative emissions for CH_4 in 2021/22 (-0.44 kg CH_4 ha^{-1}), but positive emissions in 2022/23 (0.20 kg CH_4 ha^{-1}). In 2021/22, only cumulative CH_4 emissions for F_{winter} were significantly different from CR_{winter}. In the winter of 2022/23, there were no statistical differences in cumulative CH_4 emissions between all three treatments.

Cumulative N_2O emissions were similar for CR_{winter} and FR_{winter} in 2021/22 and 2022/23 (Table 1). On average, CR_{winter} emitted 0.22 kg N_2O ha^{-1} and FR_{winter} emitted 0.36 kg N_2O ha^{-1}. F_{winter} had higher cumulative N_2O emissions (2.5 kg N_2O ha^{-1}) compared to CR_{winter} (0.18 kg N_2O ha^{-1}) and FR_{winter} (0.52 kg N_2O ha^{-1}) in 2022/23 ($P<0.05$). N_2O emissions for all treatments were generally centered around events that caused rapid changes in soil moisture, namely rains, and planned flooding and drainage.

Global warming potentials (GWP) were similar in all three treatments (CR_{winter}, FR_{winter}, and F_{winter}) in 2021/22 and 2022/23. On average, CR_{winter} had a GWP of 541 kg CO_2 eq ha^{-1}, FR_{winter} had a GWP of 295 kg CO_2 eq ha^{-1}, and F_{winter} had a GWP of 389 kg CO_2 eq ha^{-1}. For CR_{winter} and FR_{winter}, cumulative N_2O emissions made up 11% and 32% of the winter GWP on average. In contrast, the GWP of F_{winter} consisted almost exclusively of cumulative N_2O emissions.

3.4 Annual greenhouse gas emissions

Cumulative annual CH_4 emissions were greater in CR_{annual} (496 kg CH_4 ha^{-1}) than FR_{annual} (269 kg CH_4 ha^{-1}) during 2021/22 ($P<0.05$). However, in 2022/23, cumulative CH_4 emissions were similar between the two treatments (CR: 409 kg CH_4 ha^{-1}, FR: 434 kg CH_4 ha^{-1}). On average, CR_{annual} had higher CH_4 emissions (493 kg CH_4 ha^{-1}) than did FR_{annual} (325 kg CH_4 ha^{-1}) (Table 1). The summer emission period contributed to more than 96% of CH_4 emissions for CR_{annual} and FR_{annual} on average. CH_4 emissions in F_{annual} averaged -0.12 kg CH_4 ha^{-1}. As there were no CH_4 emissions in F_{summer}, F_{winter} exclusively accounted for all CH_4 emissions in F_{annual}.

Annual N_2O emissions were similar for all three treatments in 2021/22. However, in 2022/23, cumulative N_2O emissions were greater in F_{annual} (2.7 kg N_2O ha^{-1}) than CR_{annual} (0.20 kg N_2O ha^{-1}) and FR_{annual} (0.52 kg N_2O ha^{-1}) ($P<0.05$). The majority of N_2O emissions came from the winter season for all three treatments, with average contributions of 47%, 71%, and 94% for CR_{winter}, $FR_{winter,}$ and F_{winter} respectively.

On average, CH_4 emissions in summer contributed to more than 95% of annual GWP for both CR_{annual} and FR_{annual}. Given that CR_{summer} had higher CH_4 emissions on average, this translated to CR_{annual} having a higher GWP (13,937 kg CO_2 eq ha^{-1}) compared to FR_{annual} (9,236 kg CO_2 eq ha^{-1}). In contrast, the GWP of F_{annual} was mainly due to N_2O emissions in the winter. The GWP of F_{annual} is small in magnitude and was significantly lower than CR_{annual} and FR_{annual} in both seasons, averaging 413 kg CO_2 eq ha^{-1}.

3.5 Particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) fractions

Due to differences in cumulative CH_4 emissions by year for CR_{summer} and FR_{summer}, we quantified particulate organic carbon (POC) and mineral-associated organic carbon (MAOC) fractions of the soil as indicators of CH_4 emissions. POC, the labile fraction, was higher in CR_{summer} than FR_{summer} when considered across all three years, averaging 5.5 g kg^{-1} and 4.0 g kg^{-1}, respectively ($P<0.05$) (Figure 6). When separated by year, POC was higher in CR_{summer} (5.7 g kg^{-1}) than FR_{summer} (3.4 g kg^{-1}) in 2021 ($P<0.05$). Despite no statistical differences in 2022 and 2023, POC tended to be numerically greater in CR_{summer} compared to FR_{summer} in those years. There was no difference in MAOC when considered across all three years, averaging 11.5 g kg^{-1} in CR_{summer} and 10.9 g kg^{-1} in FR_{summer} (Figure 6). However, when separated by year, MAOC was greater in CR_{summer} (12.5 g kg^{-1}) than in FR_{summer} (10.7 g kg^{-1}) in 2023 ($P<0.05$). On the whole, SOC was lower in FR_{summer} than in CR_{summer} in 2023 and also numerically on average across all three years.

4. Discussion

4.1 Overview of emissions and yield

Cumulative CH_4 emissions made up the majority of GHG emissions in the rice growing season, averaging 397 kg CH_4 ha^{-1} across years and CR_{summer} and FR_{summer}. This mean was high compared to the weighted mean of 218 kg CH_4 ha^{-1} for flooded California rice systems reported in a meta-analysis (Linquist et al., 2018). Despite higher cumulative summer CH_4 emissions, the magnitude of emissions was consistent across years. Cumulative CH_4 emissions were also similar to a previous study at the same location, where the water-seeded conventional system had a cumulative emission of 446 kg CH_4 ha^{-1} (Pittelkow et al., 2014). The values obtained for FR_{summer} were also similar to the current IPCC scaling factor of 0.59 for upland exposure duration greater than 365 days under the current Tier 1 method (Ogle et al., 2019). During the winter, low cumulative CH_4 emissions were detected for CR_{winter} and FR_{winter}, averaging 12 kg CH_4 ha^{-1} across the two years – lower than the weighted mean of 79 kg CH_4 ha^{-1} reported by Linquist et al. (2018) (Table 1). However, these low emission values were in line with several studies in California (Adviento-Borbe et al., 2013; Pittelkow et al., 2014).

Cumulative N_2O emissions during the rice growing season were small and averaged 0.20 kg N_2O ha^{-1} across years and CR_{summer} and FR_{summer} - comparable to the weighted mean of 0.15 kg N_2O ha^{-1} for flooded California rice systems (Linquist et al., 2018). Soils were anaerobic during the growing season due to the continuous flood, inhibiting the aerobic nitrification process and preventing N_2O from being released as a by-product of nitrification (Patrick Jr. & Reddy, 1976). Additionally, nitrification inhibition prevents nitrate accumulation for denitrification, removing the second pathway for N_2O emissions during the continuous flood. N_2O emissions were detected during the summer when there were rapid changes to soil moisture, specifically during initial flooding and drainage at the start and end of the growing season (Figure 5). In the winter, cumulative N_2O emission averaged 0.29 kg N_2O ha^{-1} across years and CR_{summer} and FR_{winter}, and was also close to the weighted mean of 0.15 kg N_2O ha^{-1} for flooded California rice systems (Linquist et al., 2018).

Rice yields in the three years of this study averaged 11,272 kg ha^{-1}, and were in line with previous studies conducted at the same site (Carrijo et al., 2018; Perry et al., 2022). Macronutrients (nitrogen, phosphorus, and nitrogen) were applied in excess, and plots were managed with good control of pests and disease. Yields obtained in this study were comparable to regional levels (USDA, 2024). Yields in 2023 were likely slightly lower due combine harvest instead of hand harvest.

4.2 Residue incorporation method and soil carbon fractions potentially affect growing season emissions

Our hypothesis that cumulative CH_4 emissions were higher in CR_{summer} compared to FR_{summer} was supported in two out of three years – 2021 and 2023. This finding was in agreement with previous studies showing that upland rotations decreased CH_4 emissions in the following rice crop (Janz et al., 2019; Akiyama & Yagi, 2011; Weller et al., 2016). Compared to FR, CR has more residue input due to an additional cropping cycle. CR_{winter} plots were also flooded in the winter after residue incorporation. Combined, CR had increased residue inputs and longer anaerobic periods, increasing and preserving C substrates needed for CH_4 production, respectively (Wang et al., 2019). As such, the FR system was analogous to the treatment of residue removal due to reduced soil C substrates, and had been documented to reduce growing season CH_4 emissions under field conditions (Sander et al., 2014; Song et al., 2019). The exception was in 2022, where CR_{summer} and FR_{summer} had similar cumulative CH_4 emissions.

Similar cumulative CH_4 emissions between CR_{summer} and FR_{summer} in 2022 could potentially be attributed to residue incorporation by stomping, which in turn could have caused changes in soil organic carbon fractions. Stomping is a residue incorporation strategy where the rice residue is pressed into the soil with a large cage roller (Figure 3) (Blank et al., 1993). Instead of the conventional discing operation, stomping was used in October 2021 due to unusually high rains that led to field flooding (Figure 2). The pressing action of stomping encouraged better residue contact with both soil and water, enhancing residue decomposition and soil C mineralization in CR_{winter} in 2021/22 (Curtin et al., 2008; Henriksen & Breland, 2002). In 2022, there were no differences between CR_{summer} and FR_{summer} in either the POC or MAOC fractions, suggesting similar levels of soil C substrates for CH_4 production. Likely, enhanced straw breakdown in CR mineralized residue C substrates in the winter, resulting in lower than normal CR_{summer} emissions that was comparable to $FR_{summer.}$

This was the first study to evaluate the usage of POC and MAOC as indicators of CH_4 emissions in rice systems and our results suggest potential utility. Over the three years of the study, SOC was significantly higher in the CR system, with higher SOC being driven by significant increases to the POC fraction, and to a smaller degree, increases in the MAOC fraction. With the POC fraction mostly comprising plant matter and encompassing the largest active carbon pool, the significantly higher overall POC levels found in the CR system are likely directly related to an increase in residue incorporation from the CR system. In the years where CR_{summer} had higher cumulative CH_4 emissions, POC was higher in 2021 and MAOC was higher in 2023. Whilst POC always tended higher in CR_{summer} in years with treatment effects, statistical differences could not be detected in 2023. As there were only three replications in our study, there was likely insufficient statistical power to detect differences consistently across years. As such, future research should study these soil C fractions with higher levels of replication. Interestingly, CH_4 emissions during winter were not increased by stomping – suggesting that stomping can be a strategy for residue incorporation while lowering CH_4 emissions in the next season. More importantly, these results suggest residue incorporation may affect the following season's CH_4 emission levels to a greater extent than field histories and C inputs levels, representing a knowledge gap for future research. This research question is timely especially in the face of erratic precipitation patterns during harvest, which necessitates stomping over other residue incorporation methods in California.

4.3 Summer and winter contributions to annual emissions in continuous rice (CR) and fallow rice (FR)

Cumulative CH_4 emissions during the summer contributed the most to the global warming potentials (GWP) in CR and FR. On average, summer GHG emissions (CH_4 and N_2O) accounted for more than 96% of the GWPs in CR_{annual} and FR_{annual}, of which cumulative CH_4 emissions accounted for more than 99% of the GWPs in both CR_{summer} and FR_{summer}. Our results show that annual emissions from these two systems are strongly driven by CH_4 emissions during the summer season and were consistent with previous research performed in the region (Adviento-Borbe et al., 2013; Pittelkow et al., 2014). On the other hand, low N_2O emissions contributed an almost negligible amount to GWP in the summer, which is characteristic of California rice systems (2018).

Winter GHG emissions, on the other hand, accounted for less than 4% of the GWP of CR_{annual} and FR_{annual}. Cumulative CH_4 emissions during CR_{winter} and FR_{winter} contributed to 3% and 2% of annual CH_4 emissions respectively. Linquist et al. (2018) reported that 26.5% of annual CH_4 emissions came from the winter season in a meta-analysis of California. However, winter CH_4 emission values can be extremely variable. For example, Pittelkow et al. (2014) reported that winter CH_4 emissions accounted for less than 5% of annual CH_4 emissions in California. In our study, fields were flooded late in the season, approximately during December due to climate and operational issues – rains during harvest in October 2021, and winter droughts during 2022/23. The late timing of flooding may have caused low rates of methanogenesis as soil temperatures were already cold, resulting in low CH_4 emissions. (Westermann, 1993). N_2O emissions during the winter were low and similar to previously reported values, where emissions events were detected during rapid changes in soil moisture (i.e. rains, drainages, flooding) (Linquist et al. 2018).

4.4 Summer and winter contributions to annual emissions in fallow (F)

GWP of F_{annual} was low, averaging 413 kg CO_2 eq ha^{-1}. F_{summer} accounted for 6% of the annual GWP on average. Soils in F_{summer} were completely dry, leading to negligible amounts of the microbial processes that are responsible for CH_4 and N_2O emissions (Ewing et al., 2007). N_2O emissions during F_{summer} were only associated with storm events and occurred only once after a rain event in September 2022.

In contrast, F_{winter} accounted for 94% of the annual GWP on average, and was driven mainly by N_2O emissions. Cumulative N_2O emissions was higher in winter 2022/23 (2.5 kg N_2O ha^{-1}) compared to winter 2021/23 (0.46 kg N_2O ha^{-1}). Likely, this was due to high rainfall levels during winter 2022/23 that led to high soil moisture levels, driving nitrification and denitrification processes that contribute to N_2O emissions (George et al., 1993; Linquist et al., 2011) (Figure 2). Higher levels of mineral nitrogen were also likely present in F_{winter} soils compared to CR_{winter} and FR_{winter} soils due to the dry summer conditions, leading to higher winter N_2O emissions (Ewing et al., 2007).

Interestingly, F_{winter} was a net CH_4 sink in 2021/22, but a net emitter in 2022/23 (Table 1). This behavior can also be attributed to precipitation patterns, where there was low rainfall in 2021/22 and high rainfall in 2022/23. Net CH_4 consumption is commonly observed in grassland systems with intermediate levels of soil moisture that do not reach saturation (van den Pol-van Dasselaar et al., 1998; Yue et al., 2016). Additionally, Ke et al. (2013) showed that during drainage, methanotrophy greatly increased in the bulk soils of rice fields. Likely, a similar process occurred for our plots, whereby natural drainage after the exceptionally heavy precipitation in winter 2021/22 stimulated methanotrophy, and the subsequent intermediate moisture levels sustained CH_4 consumption. In 2022/23, high rainfall from Nov 2022 to Mar 2023 likely caused soils to be anaerobic as there was standing water in F_{winter} plots, promoting methanogenesis and causing net CH_4 emissions.

4.5 Considerations for net system emissions

Our study is one of the few studies that account for GHG emissions during both the summer growing (CR_{summer} and FR_{summer}) and winter (CR_{summer} and FR_{summer}) seasons. Additionally, we also fully quantified emissions during the fallow season (F_{summer} and F_{winter}). However, a key limitation was that our work only examined CH_4 and N_2O emissions but did not fully account for annual changes to soil organic carbon. We did not follow any sets of plots for the full two-year period through the fallow (F) and fallow rice (FR) phases. Consequently, our dataset does not permit us to draw strong conclusions on potential SOC changes due to the introduction of fallow. Our work, however, does suggest that fallow likely drove down soil C stocks, and this was statistically better expressed in POC but was also visible in the slower cycling MAOC fraction. Previous work suggested that net system emission estimations can be greatly reduced if changes in SOC were accounted for (Kelley et al., 2024; Zhang et al., 2024). Utilizing change in SOC and GHG emissions to calculate net system emissions, a two-year CR_{annual} cycle has net system emissions of 27874 kg CO_2 eq ha^{-1} while F_{annual} and FR_{annual} (two year) will have a net emission value of 14892 kg CO_2 eq ha^{-1}. The inclusion of soil C change in evaluating net system emissions is pivotal as reductions in CH_4 emissions, especially upland rotations, likely come at the cost of soil C reductions. To fully account for system-level C balances, future empirical studies investigating changes in GHG emissions due to upland rotations or extended aerations in rice systems can consider including SOC change to evaluate net system emissions.

5. Conclusion

Our results demonstrated that on average, annual global warming potential (GWP) was lower in FR_{annual} (9,236 kg CO_2 eq ha^{-1}) compared to CR_{annual} (13,937 kg CO_2 eq ha^{-1}). The decrease was driven by summer CH_4 emissions, where FR had 45-53% lower cumulative CH_4 emissions than CR in two out of three years. F_{annual} had a low GWP of 413 kg CO_2 eq ha^{-1}, with winter N_2O emissions being the main contributor. This was also the first empirical study to evaluate POC and MAOC as indicators of CH_4 emissions in rice. During 2022 where CH_4 emissions were similar in FR_{summer} and CR_{summer}, there were no differences in POC and MAOC between the two systems, preliminarily demonstrating that soil C fractions in rice were sensitive to management and are potential drivers of CH_4 emissions. Future studies can further investigate the roles of C fractions, ideally with higher numbers of biological replicates. Additionally, residue was incorporated by stomping in winter 2021/22, resulting in better soil-water-residue contact, higher C mineralization, and potentially resulting in similar levels of POC and MAOC. Different residue incorporation and winter vegetation management strategies may affect soil C levels and CH_4 emissions in the next rice season, representing a knowledge gap for future research. With precipitation patterns becoming increasingly erratic in California due to climate change, fallow acreage in a system previously characterized by rice monoculture will continue to increase. By quantifying emissions of the three systems, continuous rice, fallow rice, and fallow, our results provide a valuable resource for emission estimations.

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Conflict of Interest

The authors declare no conflict of interest.

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Figures and Tables

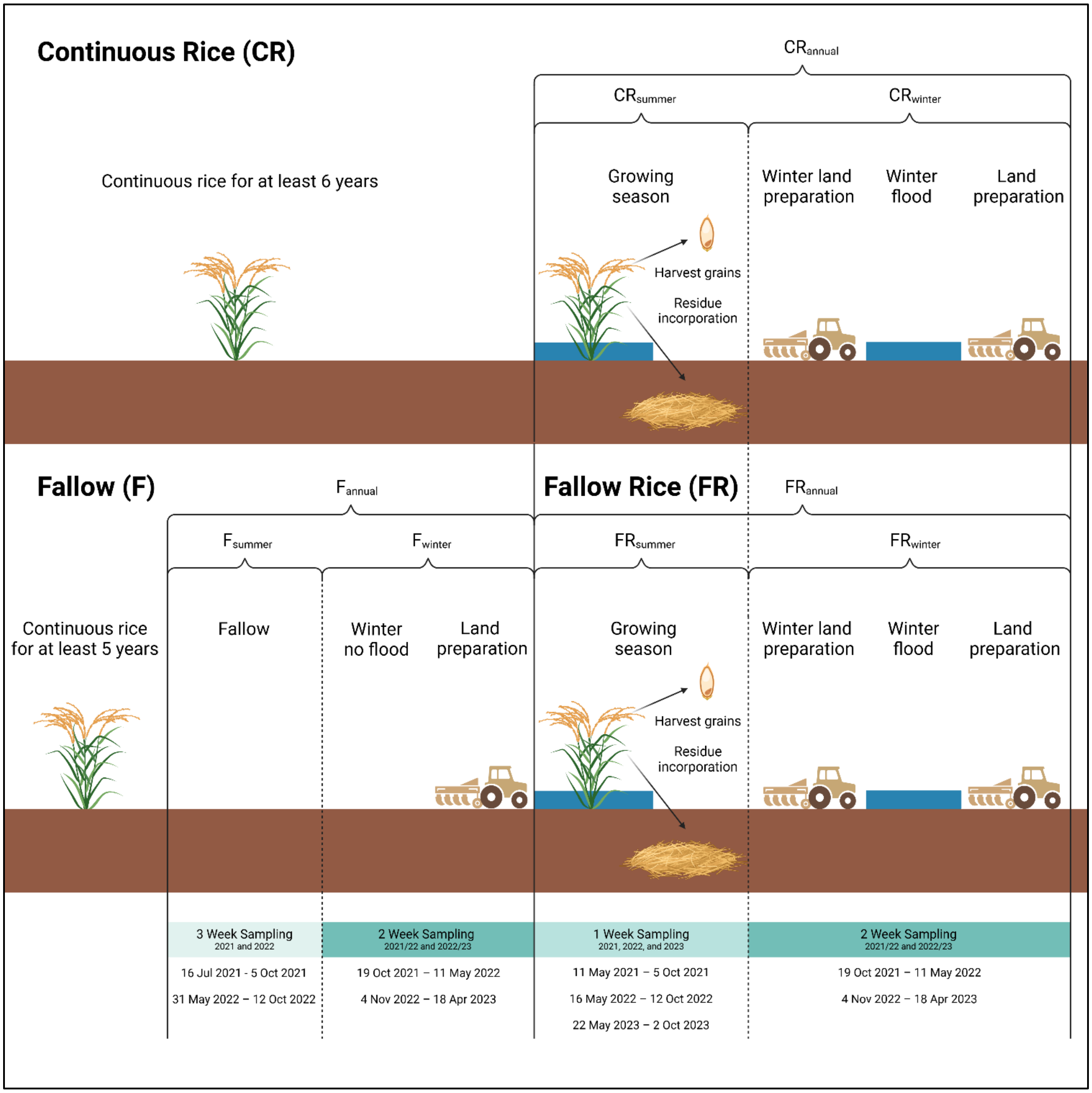

Figure 1. Overall framework of treatments, emission periods, and gas sampling intervals. All treatments were separated into summer, winter, and annual emission periods. The years in which data were collected and the associated gas sampling interval for the given emission period

are indicated in the green boxes. For each emission period, the specific start and end dates are listed at the bottom of the figure. Blue rectangles represent flooded periods.

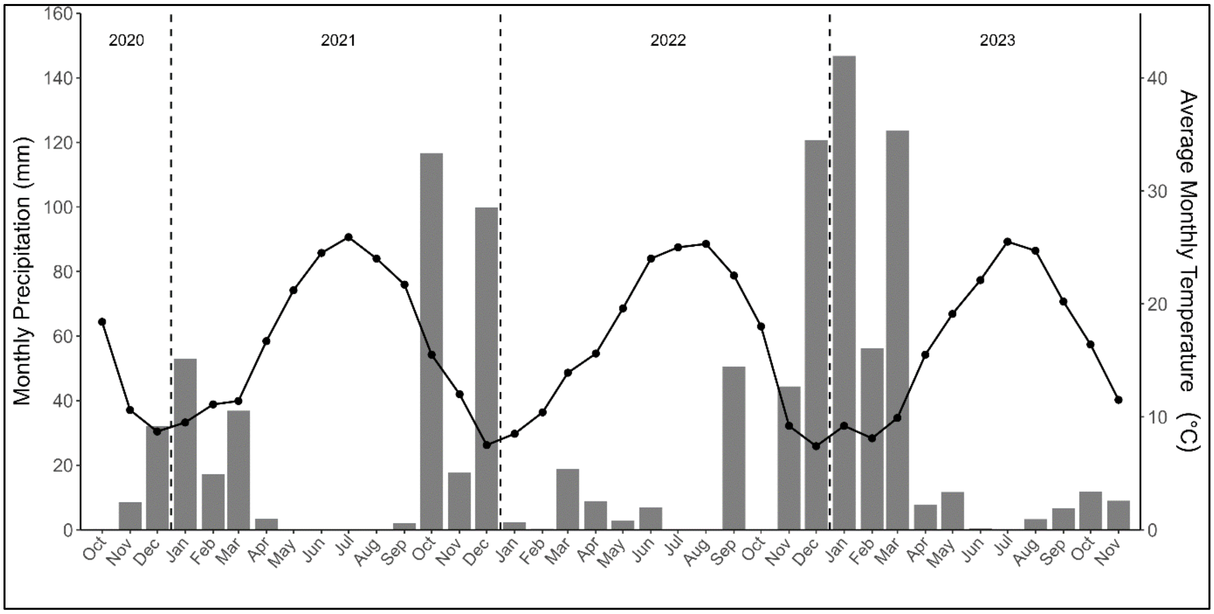

Figure 2. Monthly precipitation (grey bars) and average monthly temperature (black line) from October 2020 to November 2023 for the study site. Climate data was retrieved from the California Irrigation Management Information System (CIMIS), Biggs station.

Figure 3. Photos of machinery used for residue incorporation in this study, including the (a) disc and (b) stomper. Note that following residue incorporation with the disc, fields are flooded.

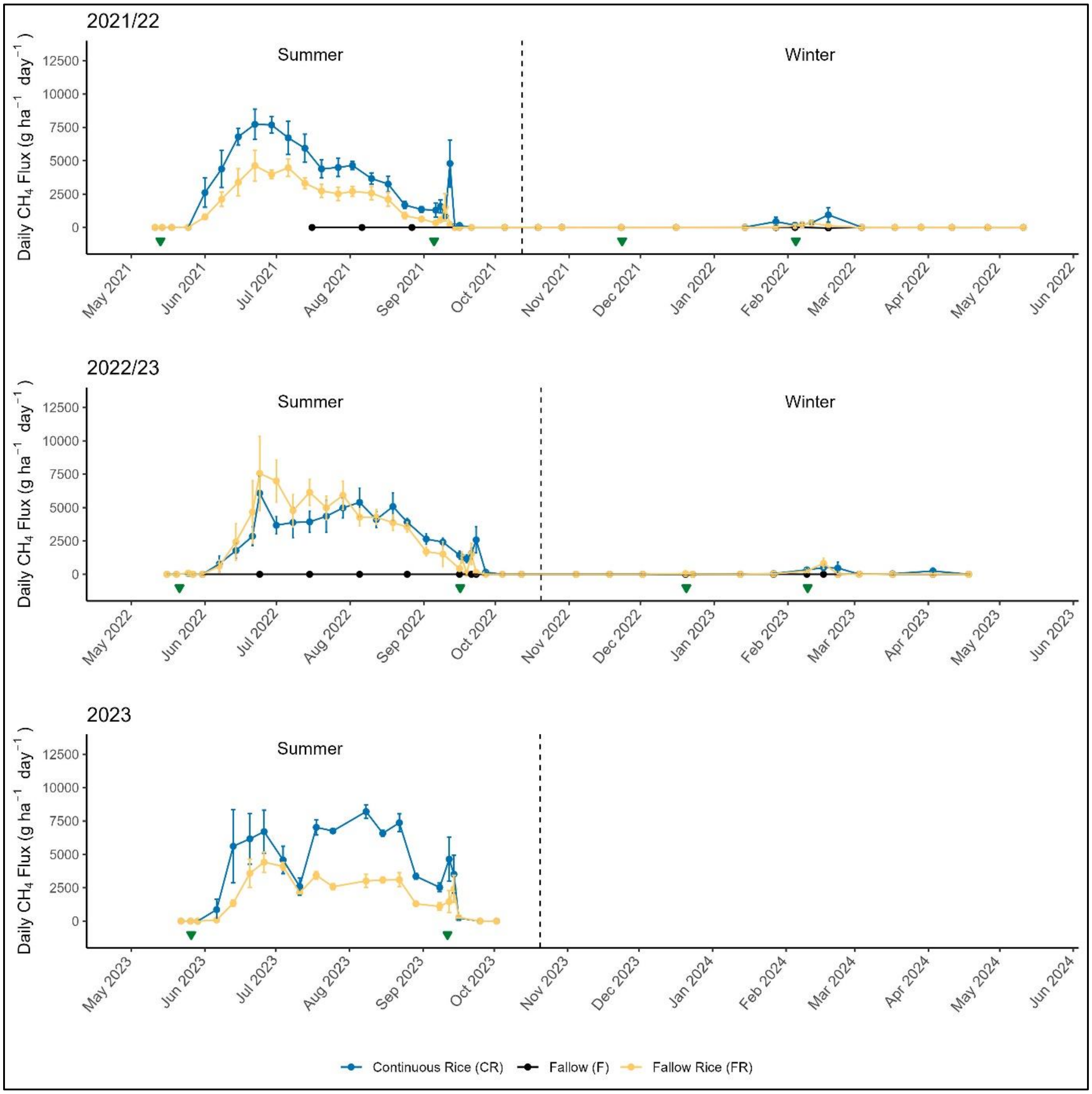

Figure 4. Daily methane (CH_4) flux during summer and winter emission periods. Error bars represent the standard error. Triangles indicate flooding and drainage events for continuous rice (CR) and fallow rice (FR) treatments only.

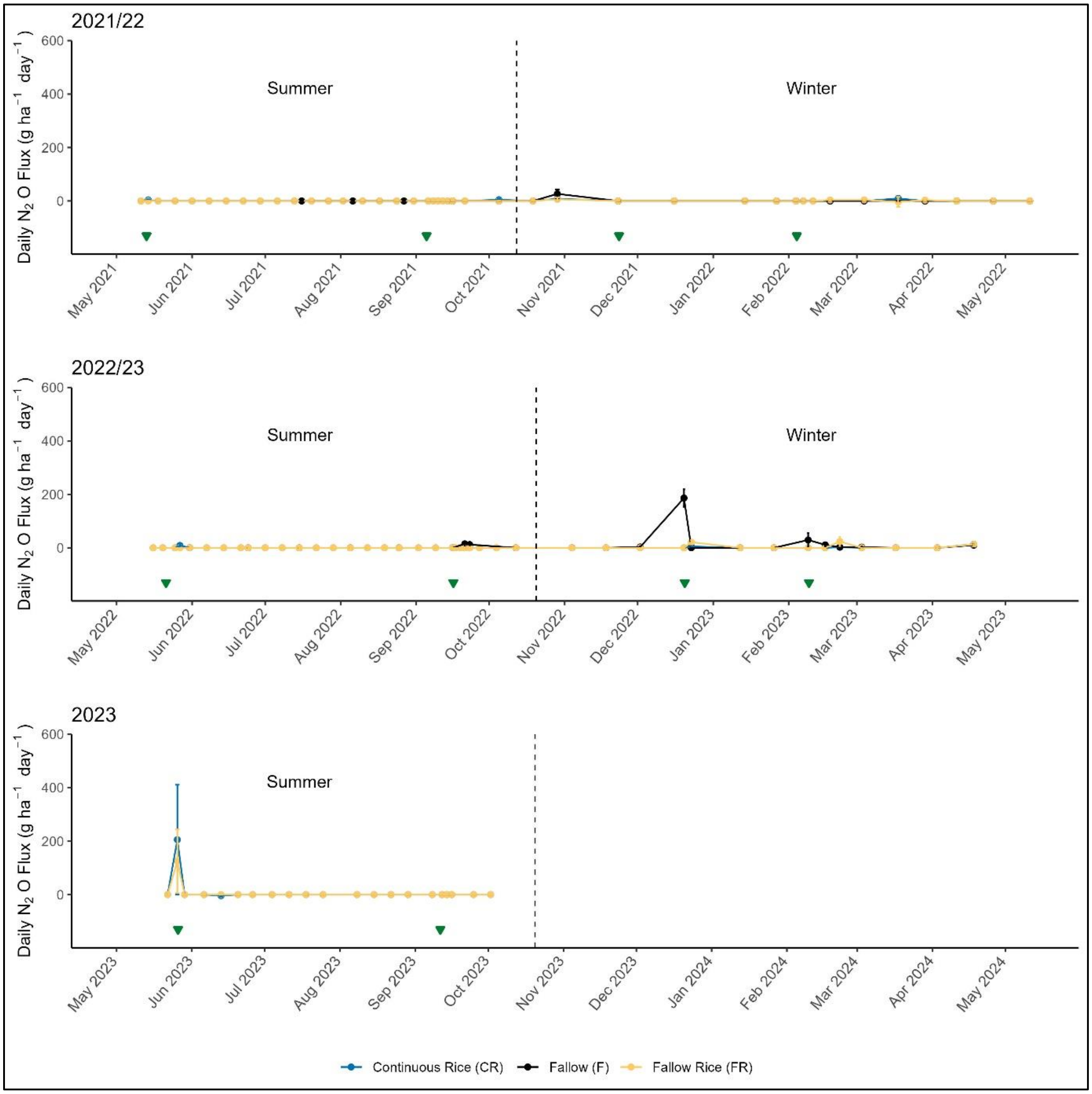

Figure 5. Daily nitrous oxide (N_2O) flux during summer and winter emission periods. Error bars represent the standard error. Triangles indicate flooding and drainage events for continuous rice (CR) and fallow rice (FR) treatments only.

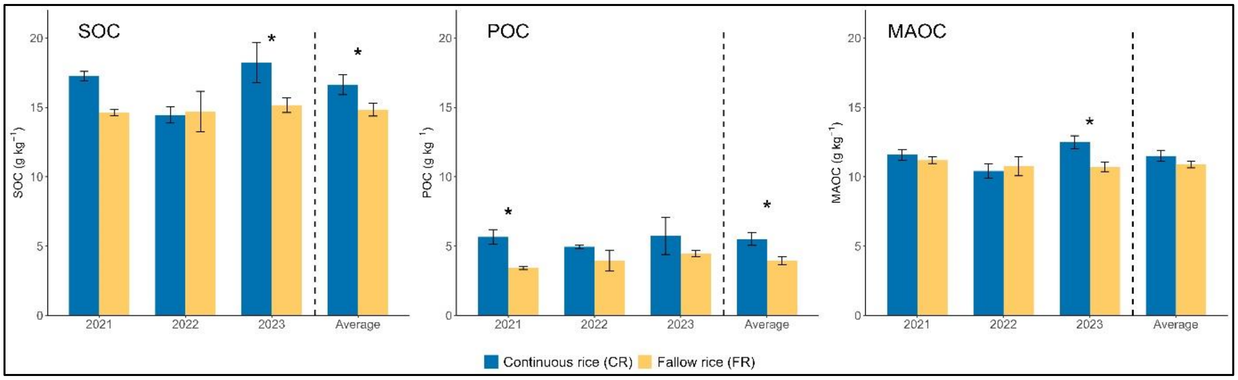

Figure 6. Soil organic carbon (SOC), particulate organic carbon (POC), and mineral-associated organic carbon (MAOC) levels in continuous rice (CR) and fallow rice (FR) soils (0-15 cm) before each summer rice season. An asterisk indicates a significant difference (P<0.05) between CR and FR. Error bars represent the standard error.

Table 1: Cumulative seasonal CH_4 emissions, N_2O emissions, global warming potential (GWP), and grain yield of the three treatments: continuous rice (CR), fallow rice (FR), and fallow (F). For values of the same response variable in a given emission period (summer, winter, or annual) during a specific year, numbers followed by the same letter are not significantly different ($P < 0.05$).

Treatment	Summer				Winter			Annual		
	CH_4 (kg CH_4 ha^{-1})	N_2O (kg N_2O ha^{-1})	GWP (kg CO_2 eq ha^{-1})	Grain yield (kg ha^{-1})	CH_4 (kg CH_4 ha^{-1})	N_2O (kg N_2O ha^{-1})	GWP (kg CO_2 eq ha^{-1})	CH_4 (kg CH_4 ha^{-1})	N_2O (kg N_2O ha^{-1})	GWP (kg CO_2 eq ha^{-1})
	2021				2021/22			2021/22		
CR	477a	0.05a	13360a	13040a	19a	0.26a	602a	496a	0.31a	13962a
FR	264b	0a	7382b	13845a	5.0ab	0.21a	194a	269b	0.21a	7576b
F	0c	0a	0c	-	-0.44b	0.46a	111a	-0.44c	0.46a	111c
	2022				2022/23			2022/23		
CR	394a	0.02a	11031a	12311a	15a	0.18a	479a	409a	0.20a	11510a
FR	425a	0a	11897a	11893a	9.2a	0.52a	396a	434a	0.52a	12294a
F	0b	0.18a	47b	-	0.20a	2.5b	668a	0.20b	2.7b	715b
	2023									
CR	558a	0.69a	15799a	9308a	-	-	-	-	-	-
FR	265b	0.44a	7544b	7233b	-	-	-	-	-	-
	Average									
CR	476a	0.25a	13396a	11553a	17a	0.22a	541a	493	0.47	13937
FR	318b	0.15a	8941b	10990a	7.1ab	0.36a	295a	325	0.51	9236
F	0c	0.09a	23c	-	-0.12b	1.5b	389a	-0.12	1.6	413

Chapter Four

Title

Agronomic performance and nitrogen management of continuous rice systems exposed to a year-long fallow

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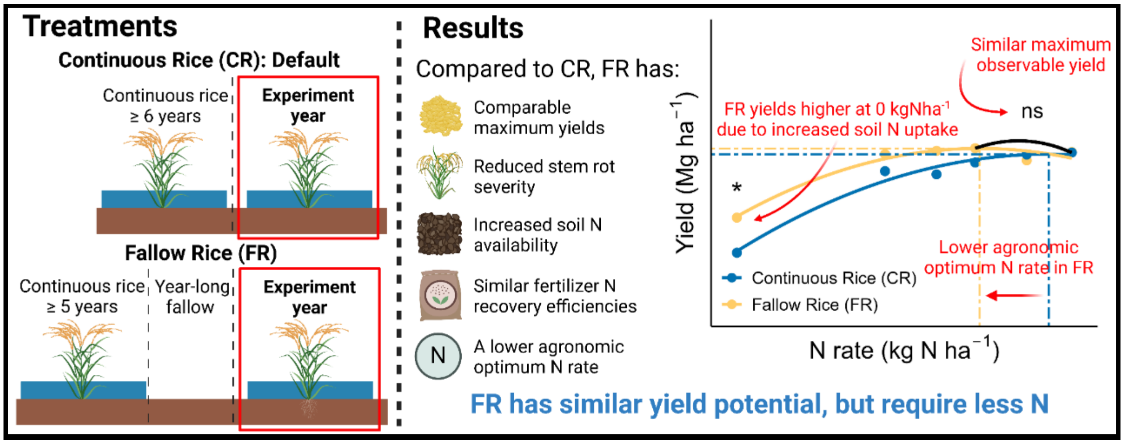
*Figures and tables are located at the end of Chapter Four.

Abstract

Erratic precipitation events including winter droughts and spring rains are increasing in California, challenging the feasibility of continuous rice mono-cropping and increasing fallows. Exposure of continuous rice soils to extended aerobic periods increases soil nitrogen (N) availability, but region-specific agronomic guidelines have yet to be developed. Yield response to N fertilization and stem rot severity were evaluated in a three-year field study for two treatments - continuous rice (CR) and fallow rice (FR – rice following a year-long fallow). Maximum observed yields did not differ between treatments, averaging 14.0 Mg ha^{-1} in 2021, 12.6 Mg ha^{-1} in 2022, and 9.6 Mg ha^{-1} in 2023. Based on quadratic regressions of yield response to N, the agronomic optimum N rate (AONR) was higher for CR in all years. Where no fertilizer N was applied, FR yielded higher than CR, averaging a difference of 2.9 Mg ha^{-1}. The yield differences at 0 kg N ha^{-1} can be attributed to soil N availability, where FR averaged 31.6 kg N ha^{-1} more soil N uptake than CR at maturity. Apparent fertilizer nitrogen recovery efficiency (FNRE) did not differ between treatments and averaged 59.8%. Stem rot, caused by *Sclerotium oryzae*, was more severe in CR than in FR, having averaged severity indexes of 3.7 and 3.1 respectively. Based on differences in soil N uptake and FNRE, N rate can be reduced by roughly 50 kg N ha^{-1} for fields following a fallow compared to continuously cropped fields, allowing growers to maintain yields with lower inputs.

Graphical abstract



Keywords

Fallow, continuous rice, yield, disease, stem rot, nitrogen management, nitrogen use efficiency

Abbreviations

FR – fallow rice

CR – continuous rice

FNRE – fertilizer nitrogen recovery efficiency

AONR - Agronomic optimum nitrogen rate

Highlights

1. Fallow rice (FR) and continuous rice (CR) were evaluated for yield response to N.
2. Soil N uptake in the zero N treatment was 31.6 kg N ha^{-1} greater in FR than in CR.
3. Maximum observed yields were similar, but the optimum N rate was lower in FR.
4. Stem rot severity was higher in CR.
5. N rate can be reduced by around 50 kg N ha^{-1} in FR while maintaining yields.

1. Introduction

Erratic precipitation patterns are causing increased fallow frequencies in California rice production. Winter droughts reduce surface water availability, leading to restrictions from irrigation districts in the next growing season (Funk et al., 2014; Gebremichael et al., 2021; Keppen & Dutcher, 2015). Heavy spring rains occurring during April and May disrupt field operations, driving growers to opt for "Prevented Planting" insurance instead of planting a crop (USDA-RMA, 2024). Currently, approximately 60% of California's rice fields are under continuous mono-cropping - which is considered the default cropping system (Salvato et al., 2024). With increased disruptions from erratic precipitation, farmers are more likely to interject year-long fallows into these continuous rice systems rather than growing an alternative upland crop, changing soil nitrogen (N) availability (Dobermann et al., 2000). Consequently, there is a need for research to evaluate the agronomic performance of rice cropping following a fallow year, especially crop fertility and pest management strategies (Rosenberg et al., 2022).

The introduction of a year-long fallow to continuous rice systems results in a prolonged aeration period, potentially leading to different nitrogen (N) cycling patterns, soil carbon quality and quantity, and necessitating adjustments to fertility management (Zhang et al., 2024, 2025). During the fallow year, fields are disced in the spring after harvest. They remain unmanaged and unflooded for a whole year before returning to rice (Figure 1) (Zhang, 2025). Winter precipitation during the year-long fallow leads to moist but unsaturated soils, increasing soil aerobic decomposition, especially recalcitrant soil organic matter that stabilizes N (Gale & Gilmour, 1988). Previous field studies have shown that rice systems following aeration periods had increased soil N availability when entering the next rice crop. Zhang et al. (2025) demonstrated that rice following a year-long fallow had 16.8 kg N ha^{-1} more N uptake from the soil pool compared to continuous rice. Similarly, Olk et al. (2009) reported that rice following upland rotations had 40 kg N ha^{-1} more soil N uptake. Despite an upland rotation being a different system and not completely analogous to a fallow, both systems share the characteristic of prolonged aeration periods. In both studies, reduced soil N uptakes in continuous rice were attributed to higher levels of soil phenols that stabilize N through mechanisms such as covalent analide bonds (Schmidt-Rohr et al., 2004).

Continuous rice systems in California have lower soil N availability compared to rice following a year-long fallow for several reasons. Firstly, continuous rice fields are flooded for approximately seven to eight months a year. Under typical management, fields are continuously flooded for the full summer growing season. After harvest in the fall, residue is incorporated into soils and flooded in the winter to aid decomposition (Linquist et al., 2006). Second, the continuous rice system has additional rice residue input compared to rice following fallow. Increased residue inputs in the continuous rice system provide additional substrates for soil phenol formation, while longer anaerobic soil conditions (summer and winter flooding) preserve phenol levels (Mahieu et al., 2000). Despite differences in soil N availability between continuous rice and rice following aerobic exposure, fertilizer N availabilities were similar. Using ^{15}N isotope tracers, approximately 30% of the applied N rate was recovered by the crop (Olk et al., 2009; Zhang et al., 2025). Similar fertilizer N availability between continuous rice and rice following aerobic exposure suggests that N deficiencies may be mitigated with higher fertilizer N rates. Consequently, the agronomic optimum N rates (AONR) may be lower in rice following aerobic exposure.

Besides differences in soil N cycling due to aeration periods, there exist other yield-reducing factors such as disease pressure. Regionally, stem rot caused by the fungus *Sclerotium oryzae* is a disease that can lead to a 22% yield loss, and severity increases with N rates (Keim & Webster, 1974; Krause & Webster, 1973). Because continuous rice systems may require a higher fertilizer N rate to maintain yields, this system may be more susceptible to stem rot. Additionally, a year-long fallow may result in reduced disease pressure. One less residue incorporation event reduces inoculum input, and the year-long fallow allows a longer time for sclerotia degradation. Together, these two factors may result in lower inoculum levels when the field exits the fallow phase. Despite no formal investigation, California rice growers have observed lower disease pressure when rice was rotated with upland crops (Rosenberg et al., 2022). A replicated field study focusing on the effect of fallow on stem rot severity can thus scientifically validate grower perceptions, allowing more effective disease management.

To address increasing fallow frequencies in California rice, a three-year field study consisting of N rate trials was conducted to evaluate crop response to N and agronomic performance of two treatments: continuous rice (CR) and fallow rice (FR – rice following a fallow). The main objective was to determine the yield potential, N use efficiency, N uptake, and agronomic optimum N rates (AONR) of the two systems. We hypothesized that the two treatments would have comparable maximum observed yields and fertilizer N recovery efficiencies (FNRE), but in FR, increased soil N uptake will result in a lower AONR. Related to yield potential, a secondary objective was to evaluate stem rot disease severity. We hypothesize that stem rot will be more severe in CR, and if so, may reduce the yield potential. Overall, these results will provide directions for fertilizer N and disease management after fallow years, allowing growers to maintain yields with optimum rates of inputs.

2. Materials and methods

2.1 Study site

Experiments were conducted at the Rice Experiment Station in Biggs, California (39°27’47” N, 121°43’35” W) from 2021 to 2023. This site has a history of rice cultivation for the last few decades. The soil was an Esquon-Neerdobe Complex, classified as fine, smectitic, thermic Xeric Epiaquerts and Duraquerts, and soil texture consisted of 290 g kg^{-1} sand, 260 g kg^{-1} silt, and 450 g kg^{-1} clay (Pittelkow et al., 2012). Other soil characteristics (0-15 cm) include pH between 5.3 - 5.7 (1:1 soil/water), cation exchange capacity (CEC) (sodium acetate) between 26.3 – 33.4 cmolc kg^{-1}, and organic matter content (Walkley-Black titration) between 23 – 39 g kg^{-1} (Midwest Laboratories, 2024). The site has a Mediterranean climate - little to no summer precipitation and high winter precipitation, averaging 90 mm and 260 mm respectively across the three study years (CIMIS, 2024).

2.2 Treatments, experimental design, and plot management

The experiment was set up as a split-plot design in a randomized complete block with three replications per year. The two main plot treatments were CR and FR (Figure 1). Each main plot was approximately 0.25 ha in size. Continuous rice refers to rice following at least six years of rice while FR refers to rice following a year-long fallow (before the fallow, rice was grown continuously for at least five years). In the year-long fallow, plots were disced in the spring and then left unmanaged until the land preparation phase of the experimental year. Main plots were managed identically during the experiment year. After the main plots dried in the spring (March to April), they were chiseled, disced, leveled, and rolled.

Within each main plot, there were six randomized subplots, representing low to high annual N rates ranging from 0 kg N ha^{-1} to 260 kg N ha^{-1} (see Figure 1 for specific N rates). Nitrogen fertilizer was applied as aqueous NH_3 to subplots after leveling. Rice seeds (variety M206) were soaked for at least 24 hours in water before direct seeding into flooded fields at a rate of 200 kg ha^{-1}. A continuous flood was maintained during the growing season and main plots were drained three weeks before harvest. Potassium (2021: 39 kg K ha^{-1}, 2022: 33 kg K ha^{-1}, 2023: 40 kg K ha^{-1}) and phosphorous (2021: 20 kg P ha^{-1}, 2022 and 2023: 22 kg P ha^{-1}) were applied at non-limiting rates. Azoxystrobin fungicide (Quadris) was applied at a rate of 0.14 kg a.i. ha^{-1} in all three years in early August during the period of panicle emergence. Other pests and weeds were controlled as per standard practice. After harvest, rice residue was incorporated and flooded in the winter to aid in decomposition (Linquist et al., 2006). For additional details on plot management, please refer to the supplementary materials.

2.3 Yield quantification

Grains were harvested at physiological maturity when grain moisture levels were below 26%. In 2021 and 2022, plants were harvested by hand from 1 m^2 harvest rings that were placed into subplots before flooding. Plants were cut right above the soil and used for yield quantification and other analyses. Harvested plants were dried at 60°C before grains were separated from straw using hand stripping to obtain yield. In 2023, subplots were harvested using an ALMACO small plot combine. All yields were adjusted to 14% moisture per convention (Fageria, 2007). We consider the maximum observed yield in each treatment to be representative of the system's yield potential or attainable yield (Fischer, 2015).

2.4 Crop nitrogen uptake and fertilizer recovery

Crop N uptake at physiological maturity was the N present in all above-ground biomass. In 2021 and 2022, plants were harvested and processed as described in section 2.3. Straw and grain weights were used to determine N uptake. In 2023, a representative subsample was taken from each subplot before the combine harvest. The subsample was oven-dried and separated into straw and grains to determine the harvest index (HI). Harvest index was used to estimate straw weights from combine harvest grain weights. In all three years, separated grains and straw were ground to a powder and weighed into tin capsules. Total N in the tin capsules was analyzed by combustion and EA-IRMS (elemental analyzer isotope ratio mass spectrometry) at the UC Davis Stable Isotope Facility (UC Davis SIF, 2023). N uptake (kg N ha^{-1}) was then calculated as the sum of N present in both grains and straw for each subplot. Apparent fertilizer N recovery efficiency (FNRE) for each subplot was calculated using equation (1),

$$(1)\ FNRE\ (\%) = \frac{N\ uptake_{fertilized} - N\ uptake_{unfertilized}}{Total\ N\ applied} \times 100$$

whereby N $uptake_{fertilized}$ and N $uptake_{unfertilized}$ subplots were in the same main plot (Kelley et al., 2024; Kongchum et al., 2024).

In 2021 and 2022, N uptake over time was evaluated for subplots with N rates of 0 kg N ha^{-1} and 185 kg N ha^{-1} (Figure 1). Plants were harvested at PI (panicle initiation) and 50% heading (H) from harvest rings placed into subplots before flooding. Plants were sampled from the rings, thoroughly rinsed to remove soil, and had roots removed. The samples were then oven-dried at 60 °C before being ground to a powder and analyzed for N uptake as detailed above.

2.5 Stem rot evaluation

Stem rot severity was quantified on a scale of 1 to 5 via visual inspection after the end-of-season drain each year (Krause & Webster, 1973). Because disease severity was reported to increase with N rate, rice plants were harvested from the highest N rate for ease of visual inspection (Martínez, 2021). Per plot, 25 tillers were sub-sampled and assigned a severity score from 1 to 5 (1 = healthy; 2 = slight infection with sclerotia and symptoms observed on the outer leaf sheath; 3 = mild infection with observed discoloration in the inner leaf sheaths, but a healthy culm; 4 = moderate infection with slight to mild discoloration of the culm; 5 = severe infection with internal infection of the culm). The disease index for each subplot was calculated using equation (2),

$$(2)\ Disease\ index\ (scale\ of\ 1\ to\ 5) = \left(\sum_{n=1}^{5} T_n \times n\right) / T_{total}$$

whereby n was the severity score, T_n was the number of tillers having the associated severity score, and T_{total} was 25 (total number of tillers evaluated for a given subplot).

2.6 Statistical analyses and data visualization

Statistical analyses and data visualization were performed in R-Studio (R Core Team, 2024). Scripts and data used for the analysis are available online (https://github.com/XiaoZhangZhangRice/FallowRice_ContinuousRice_AgronomicPerformance). Data processing was primarily performed using the “dplyr” package (Wickham et al., 2023). As this was a multi-year experiment, treatment and year effects were first investigated. Yield was fitted with treatment, N rate, and their interaction as fixed effects, and “block:treatment” and “block” as random effects using a linear mixed effects model with the “lmer” function (Bates et al., 2015). Analysis of variance (ANOVA) showed a significant year effect ($P<0.05$, Table 1), prompting us to present results by year for the majority of response variables. For each year, yield was fitted with the same terms as the above model. ANOVA was conducted, and pairwise comparisons at each N rate were made between treatments with Tukey adjustments and the “emmeans” package (Lenth, 2022). Additionally, differences in maximum observed yield between treatments were obtained with “emmeans” and Šidák adjusted p-values. Lastly, for each year, yield and N rate were fitted with quadratic regression models with the “lm” function (Cerrato & Blackmer, 1990; Watkins et al., 2010). The agronomic optimum nitrogen rates (AONR) were calculated from the local maxima of the quadratic regressions.

Nitrogen uptake at maturity and FNRE were analyzed using the same approach as yield. Additionally, N uptake at maturity and N rate were fitted with linear regression models for each experimental year. N uptake over time was evaluated to have no year effect, so results were pooled from 2021 and 2022. N uptake over time was separated by N rate (0 and 185 kg N ha^{-1}) and fitted with growth stage, treatment, year, and their interactions as fixed effects, and “block:treatment” and “block” as random effects. Lastly, for stem rot severity, treatment, year, and their interaction were fitted as fixed effects and “block” as the random effect. ANOVA and Tukey’s pairwise comparisons were conducted to determine treatment effects for N uptake at maturity, N uptake over time, and stem rot severity. Graphs, tables, and other data visualizations were created with the statistical findings using “ggplot2” and “ggpubr” packages (Kassambara, 2023; Wickham, 2016).

3. Results

3.1 Yield response to nitrogen

There was a significant effect of N rate and treatment-by-N rate interaction ($P<0.05$) (Table 1). Year significantly affected yields, but no treatment-by-year interaction was observed. In each year, the maximum observed yield was similar between CR and FR, averaging 14.0 Mg ha^{-1}, 12.6 Mg ha^{-1}, and 9.6 Mg ha^{-1} in 2021, 2022, and 2023 respectively. When no fertilizer N was applied (0 kg N ha^{-1}), FR always yielded higher than CR ($P<0.05$) (Figure 2). Averaged across all experimental years, the yield in FR (7.8 Mg ha^{-1}) was 59% higher than CR (4.6 Mg ha^{-1}) at 0 kg N ha^{-1}.

Fallow rice had a lower agronomic optimum N rate (AONR) compared to CR. Based on quadratic regressions, the AONR, which corresponds to local maxima, was lower in FR compared to CR in all three years, averaging 58 kg N ha^{-1} (Figure 2) The quadratic regressions of yield response to N rate generally achieved a good fit ($R^2>0.7$) except FR in 2023. These results demonstrate that maximum yields were comparable between CR and FR, and that AONR was lower in FR.

3.2 Crop nitrogen uptake

Nitrogen uptake over time at specific growth stages displayed little effect of year or year by treatment interaction. Consequently, 2021 and 2022 data were pooled for analysis. Throughout various physiological stages at panicle initiation (PI), 50% heading, and maturity, N uptake was greater in FR for N rates of 0 kg N ha^{-1} and 185 kg N ha^{-1} ($P<0.05$) (Figure 3). These results suggest that N deficiencies were apparent at PI, and persisted till physiological maturity.

Differences in N uptake at maturity between CR and FR were 28.6 kg N ha^{-1} for 0 kg N ha^{-1}, and 35.3 kg N ha^{-1} for 185 kg N ha^{-1}. Crop N uptake at maturity was also quantified for other N rates (Figure 4). The line of best fit for the linear regression of FR was always at higher N uptake levels than CR, and the 95% confidence intervals have little overlap. Overall, these results suggest that N uptake at maturity was higher in FR compared to CR across N rates in all years.

Additionally, fertilizer nitrogen recovery efficiency (FNRE) at maturity was quantified. During all experimental years, no statistical differences were detected across N rates and treatments (Table 2). These findings support our hypothesis that fertilizer N recoveries were similar between treatments. Averaged across years, treatments, and N rates, FNRE was 59.8%.

3.3 Stem rot severity

Stem rot severity was evaluated from subplots with the highest N rate (2021 and 2022: 260 kg N ha^{-1}, 2023: 210 kg N ha^{-1}). Stem rot severity was higher in CR than FR in all years ($P<0.05$), with evidence of slight year-to-year variation (Figure 5). Averaged across years, the severity index was 3.7 for CR and 3.1 for FR.

4. Discussion

4.1 Yield variation across years

Year had a large effect on yields in this study. Statewide yields were the highest in 2021 (10.1 Mg ha^{-1}), followed by 2022 (9.8 Mg ha^{-1}), and lastly 2023 (9.6 Mg ha^{-1}) (USDA, 2024). Similarly, in this study, the highest yields were observed in 2021 and the lowest in 2023 (Table 1), reflecting regional trends. Additionally, yields in 2023 may have been lower because the crop was harvested by combine, which may depress yields compared to hand harvesting (Bunna et al., 2019; Weber & Fehr, 1966).

4.2 Maximum observed yields

Maximum observed yields were similar between CR and FR where N rates were sufficient. This finding may appear to contradict previous findings that upland rotations in rice, which is similar to aerobic exposure in FR, increased yields compared to continuous rice (Linh et al., 2015; Olk et al., 2009). Where no N was applied in our study, yields were higher in FR than in CR, agreeing with previous findings that upland rotations have a yield benefit (Figure 2). However, previous studies did not utilize an N response trial, and their yield differences between rice monoculture and upland rotations were quantified at a single N rate. Given that reduced soil N availability was the main reason for yield reduction in continuous rice, it is unsurprising that similar maximum yields can be achieved by increasing fertilizer N rates. Lastly, a yield contest study in California that examined factors influencing high yields reported that the highest-yielding fields were not influenced by previous upland rotations and fallows – consistent with the findings of this study that a year-long fallow does not increase the yield potential of the system (Linquist et al., 2025).

However, the numerical value of maximum observed yields tended to be higher in FR than CR and can be partly attributed to stem rot disease severity. Averaged across years, the severity index was 3.7 for CR and 3.1 for FR, indicating that disease severity was high in both systems, but pathogen penetration into the culm occurred at a higher frequency in CR. The FR treatment did not receive residue incorporation during the year-long fallow (Figure 1). In comparison, the CR treatment received one additional season of residue incorporation before the experimental year, increasing inoculum input and subsequently, stem rot severity. Bockus et al. (1979) reported that residue removal after harvest reduced inoculum levels compared to residue incorporation, reinforcing our finding. Higher than usual stem rot severity in 2022 was likely due to the winter drought in 2021/22 (CIMIS, 2024) (Figure 2), whereby drier soil moistures were less conducive for sclerotia decomposition (Cintas & Webster, 2001; Usmani & Ghaffar, 1986). In the study, azoxystrobin pesticide was used to reduce control stem rot severity. Because all plots were treated, it was not possible to determine the extent of yield loss due to the disease. Nevertheless, higher severity levels in CR compared to FR plots indicate that stem rot may contribute to the maximum yield difference between them (Maschmann et al., 2010). This was the first study in California to examine the effect of fallowing on stem rot severity, experimentally validating growers' perception of lower disease pressure with rotations, even with fungicide applications (Rosenberg et al., 2022).

4.3 Agronomic optimum N rates and management recommendations

Agronomic optimum N rates were higher in CR than FR when determined with quadratic regressions. Higher AONR in CR was driven by reduced soil N uptake, which was best captured in the 0 kg N ha^{-1} subplots. Averaging across all three years, CR had 31.6 kg N ha^{-1} less N uptake than FR at maturity (Figure 4). The difference in soil N uptake was already observable at panicle initiation between CR and FR, and persisted until physiological maturity (Figure 3). Quadratic regression of yield response to N is a well-established method in rice systems to determine AONR (Watkins et al., 2010). Generally, the quadratic models provided a good fit (R^2>0.7) except FR in 2023 (R^2 = 0.32). Additionally, there was year-year variability in AONR differences, ranging from 5 kg N ha^{-1} to 115 kg N ha^{-1}. Inconsistent AONR differences were expected due to the strong year effect on yields, resulting in shifting AONRs for both treatments (Table 1). Consequently, AONR difference was not a stable metric for calculating N rate reductions.

Reduced soil N availability but similar FNRE in CR was wholly consistent with previous findings. Zhang et al. (2025) and Olk et al. (2009) both demonstrated using ^{15}N tracers that soil N was more available in rice following aerobic exposure compared to continuous rice systems. In this study, N uptake at maturity for 0 kg N ha^{-1}, which represents indigenous soil N uptake (Cassman et al., 1996), was lower in CR (Figure 3). Reduced soil N availability can be attributed to increased levels of soil phenols in rice monoculture (the CR treatment in this study) that stabilize N through mechanisms such as covalent bonding (Schmidt-Rohr et al., 2004). However, reduced soil N availability in CR did not result in reduced maximum observed yield due to similar FNRE between treatments. In our study, FNRE was determined with a relative method using a 0 kg N ha^{-1} reference. FNRE did not differ between CR and FR, averaging 59.8%. Similarly, fertilizer N recoveries also did not differ between continuous rice and rice following aerobic exposure when determined by ^{15}N isotopes (Olk et al., 2009; Zhang et al., 2025). However, the recovery efficiency obtained using ^{15}N was numerically lower at around 30%. Consequently, soil N deficiencies in CR can be corrected with higher fertilizer N rates, removing N limitations on yield, but necessitating higher N rates to achieve yield potential (Cassman et al., 1995). For a full discussion of N cycling patterns between continuous and rice following aerobic exposure, the reader is referred to previously published literature on this topic (Olk et al., 2009; Schmidt-Rohr et al., 2004; Zhang et al., 2025).

For a rice field exiting a year-long fallow, it is clear that similar yields to CR can be achieved at a lower N rate. However, there are different ways to estimate optimal N rate reductions. In our study, the year had a strong effect on yields, possibly due to climatic factors such as temperature, causing AONR differences between treatments to vary across years (Table 1 and Figure 2). Comparatively, a more stable metric was differences in soil N uptake in 0 kg N ha^{-1} subplots. Across years, the differences in soil N uptake were generally consistent, ranging from 25.6 kg N ha^{-1} to 37.5 kg N ha^{-1} and averaging 31.6 kg N ha^{-1}. Using an average FNRE of 59.8% found in this study (Table 2), the N rate in FR can be reduced by 52.8 kg N ha^{-1} compared to CR without a N uptake penalty. This value is close to the reduction of 52 kg N ha^{-1} reported by Zhang et al. (2025), which was determined using ^{15}N fertilizer - a direct and robust measurement of fertilizer N recovery efficiency. Due to the congruencies of values attained by Zhang et al. (2025) and our current study, we recommend reducing preplant fertilizer N rates by around 50 kg N ha^{-1} when returning to rice after a year-long fallow. Assuming an input cost of 0.63 USD per kg N using aqua NH_3 fertilizer (Harrell et al., 2023), fertilizer N input costs are lowered by 31 USD ha^{-1} to achieve the same yield. Growers can continue in-season monitoring of crop N uptake using remote sensing tools or tissue testing, and apply topdress N midseason should any signs of N deficiencies be detected in FR (Linquist et al., 2009; Rehman et al., 2023). Given the global unpredictability of supply chains, increasing prices of N fertilizers, and increasing fallow frequencies, reducing N rates without yield penalty can help optimize inputs and enhance the financial viability of growing operations (Adjesiwor & Islam, 2016).

5. Conclusion

Our study showed fallow rice (FR) can achieve comparable yields to continuous rice (CR) at a lower N rate. Agronomic optimum N rates (AONR) were lower in FR than CR when determined by quadratic regression, while maximum observed yields were similar between treatments. Additionally, stem rot disease severity was lower in FR, averaging 0.6 less on the severity index. When no fertilizer N was applied, CR had 31.6 kg N ha^{-1} less soil N uptake at maturity compared to FR, resulting in a yield difference of 2.9 Mg ha^{-1} averaged across years. In our study, continuous rice occurred for five years before entering the year-long fallow and providing an N benefit in FR. The number of years that continuous rice monoculture needs to take place before soil N deficiencies can be observed may be shorter, representing an unaddressed knowledge gap. Fertilizer N recovery efficiencies (FNRE) were similar between treatments, averaging 59.8%. Overall, we conclude that growers can apply approximately 50 kg N ha^{-1} less fertilizer N in FR to maintain yields based on differences in soil N uptake in 0 kg N ha^{-1} subplots. The N fertilization reductions amount to cost savings of approximately 31 USD ha^{-1}.

California is a high-yielding agricultural region due to important agronomic advantages provisioned by dry summers, including full irrigation control and reduced disease pressure.

However, the viability of rice cultivation is contingent on precipitation patterns that are becoming increasingly erratic and directly challenge the norm of continuous rice. Furthermore, increased global supply chain disruptions in recent years have resulted in increased fertilizer N prices. Our work provides pragmatic N management recommendations to growers for fields exiting fallows to reduce input costs, allowing for sustained productivity and financial viability of their growing operations.

Supplementary material

Additional details of the field study are available in the supplementary material.

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Conflict of Interest

All authors declare no conflicts of interest.

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Figures and tables

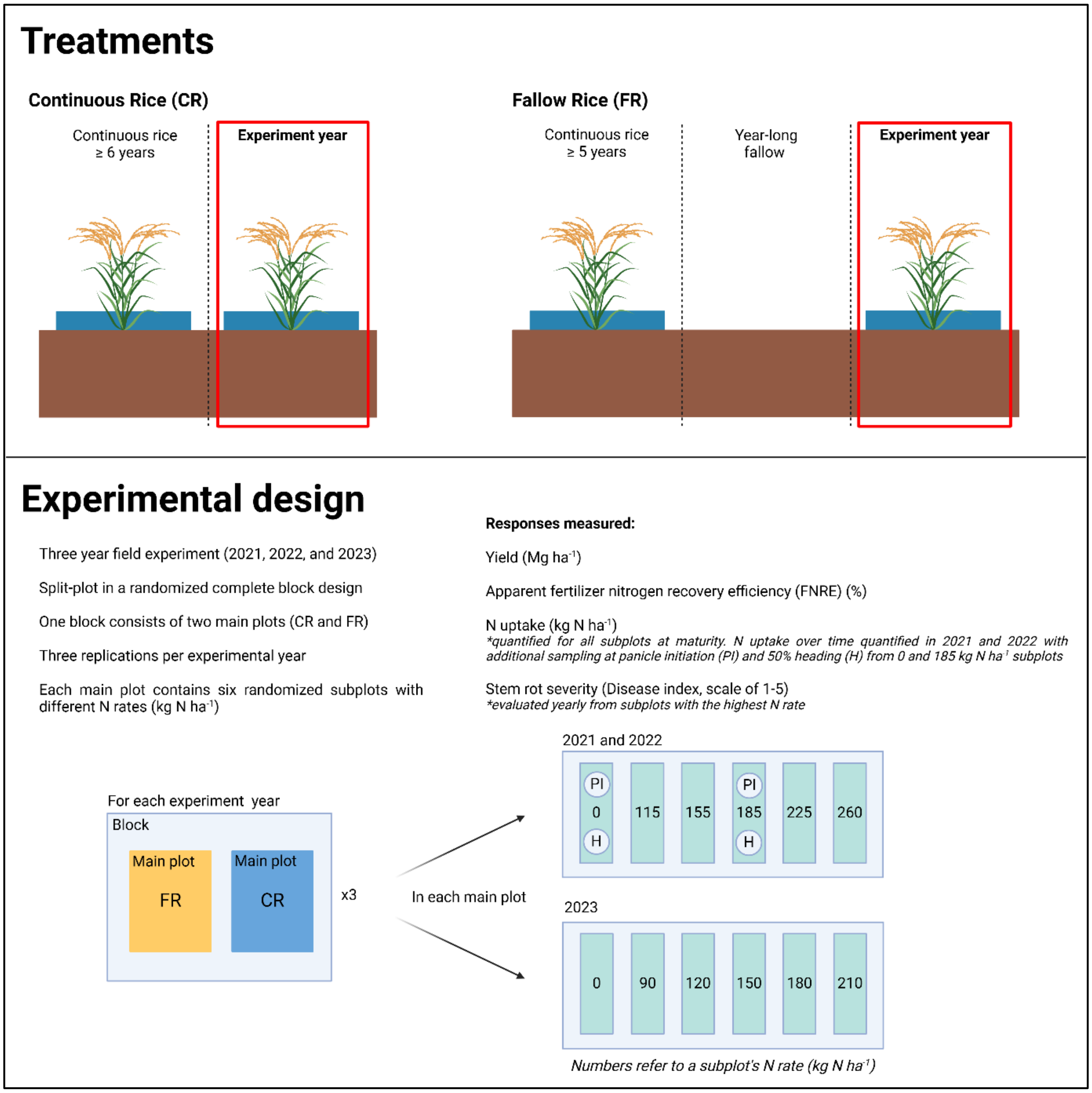

Figure 1. Visual outline of treatments, nitrogen (N) rates, and experimental design.

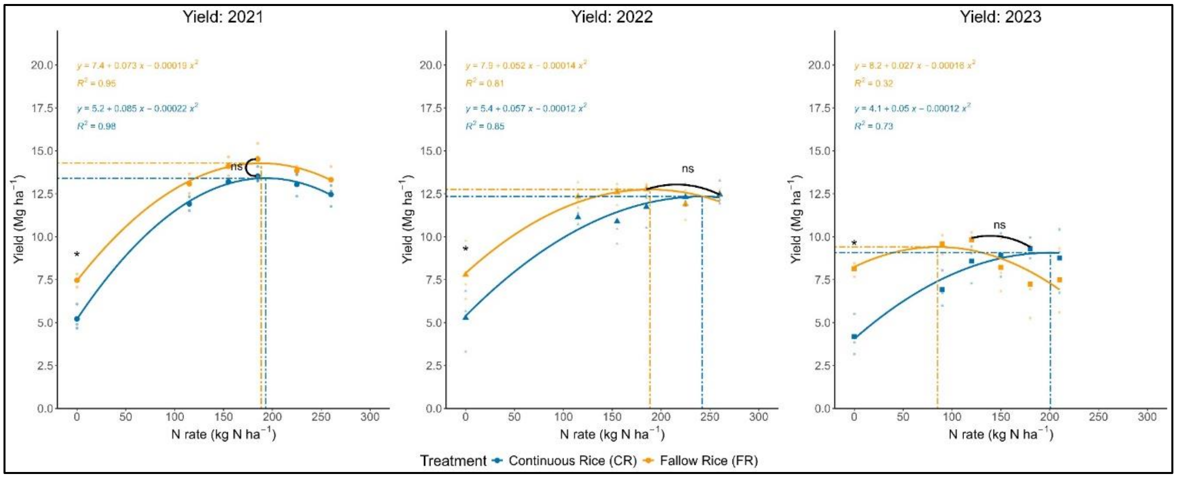

Figure 2: Yield response to nitrogen (N) rates for each of the three years. Dark points represent the treatment mean at each N rate and light points represent individual observations. An asterisk indicates a significant difference between treatments at 0 kg N ha^{-1} ($P<0.05$; Tukey's HSD). During all experimental years, the maximum observed yields between treatments were not significantly different (ns; Siddak adjusted comparisons). Results were fitted with quadratic regression models. Dashed lines indicate local maxima corresponding to the agronomic optimum nitrogen rate (AONR).

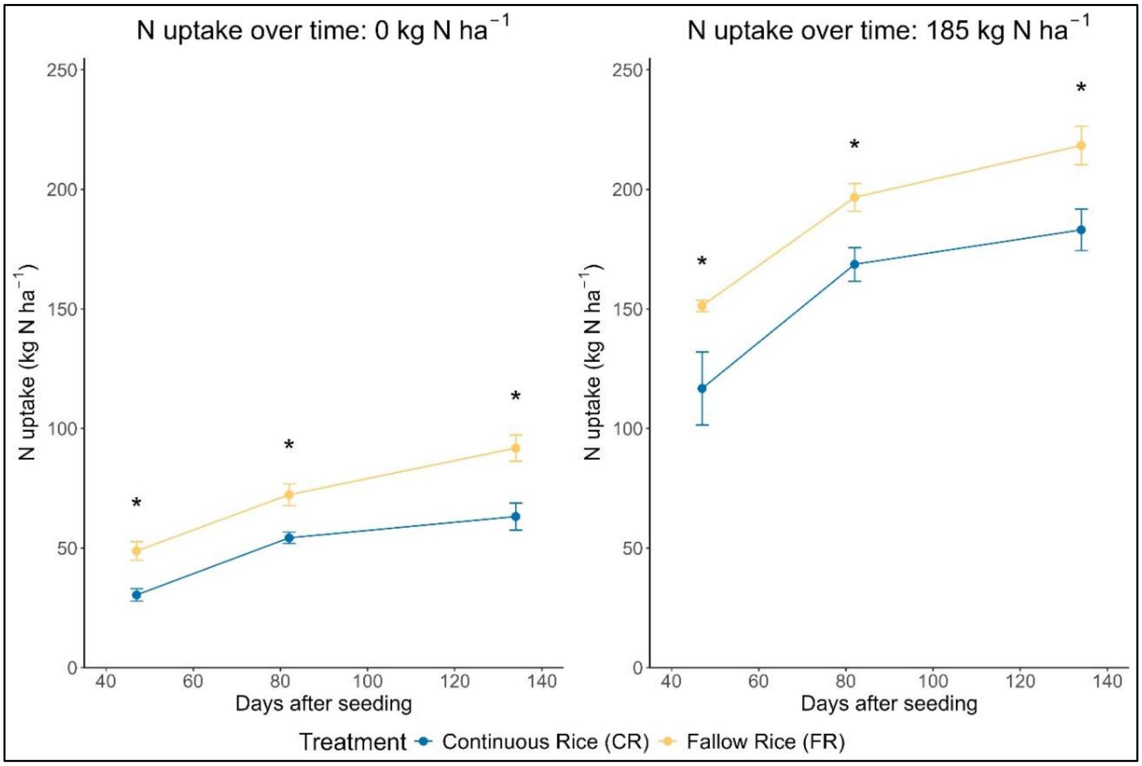

Figure 3: Nitrogen (N) uptake over time at panicle initiation, 50% heading, and physiological maturity in chronological order for two N rates (0 and 185 kg N ha^{-1}). Results were pooled from 2021 and 2022. An asterisk indicates a significant difference between treatments ($P<0.05$). Error bars represent the standard error.

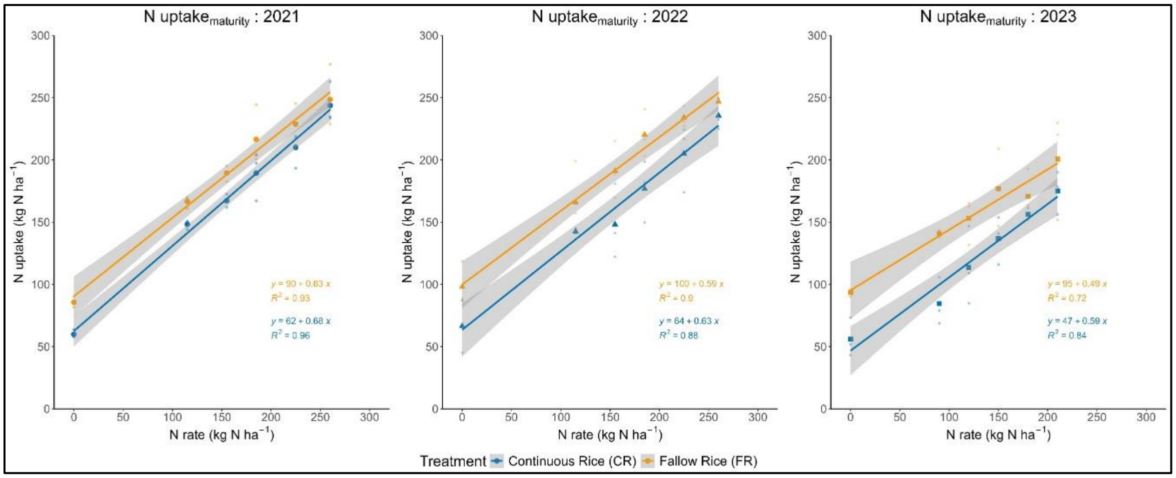

Figure 4: Crop nitrogen (N) uptake at maturity from low to high N rates for each of the three years. Results were fitted with linear regression. Dark points represent the mean at each N rate and light points represent individual observations. Shaded areas represent the 95% confidence interval.

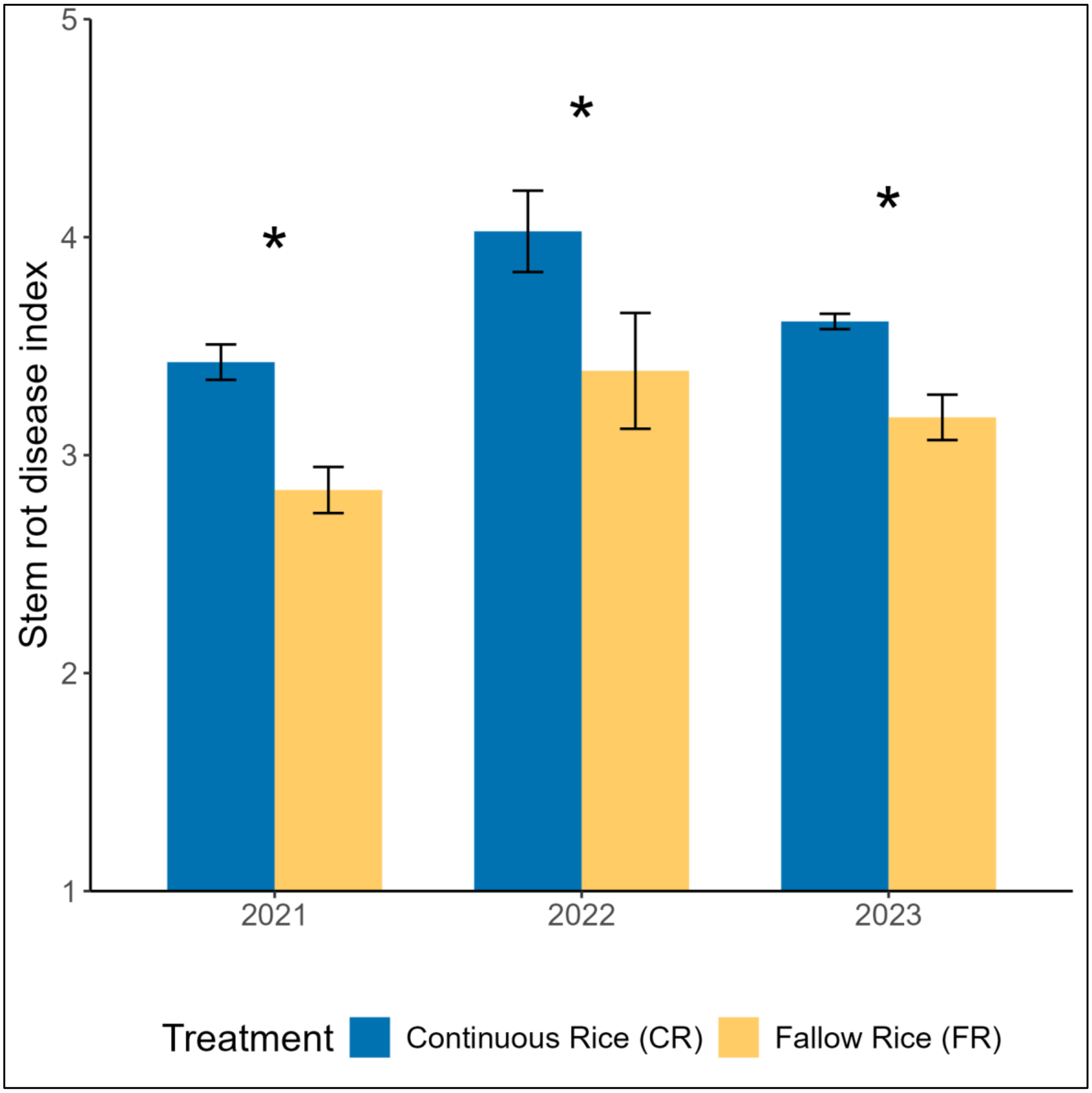

Figure 5: Stem rot severity for each of the three years. An asterisk indicates a significant difference between treatments (P<0.05). Error bars represent the standard error.

Table 1: ANOVA table showing effects of treatment, nitrogen (N) rate, year, and their interactions on yield. Average yields across all N rates for a given treatment, continuous rice (CR) or fallow rice (FR), are summarized by year.

Year		Treatment	Yield (Mg ha^{-1})
2021		CR	11.6
		FR	12.7
2022		CR	10.6
		FR	11.6
2023		CR	7.78
		FR	8.41
Source of variation	**df**	**F-value**	**P-value**
Treatment (Main plot)	1	1.9	0.19
N rate (subplot)	10	44.9	<0.01
Year	2	4.8	0.03
Treatment x Year	2	1.2	0.33
Treatment x N rate	10	5.4	<0.01

Table 2. Apparent fertilizer nitrogen recovery efficiency (FNRE) of continuous rice (CR) and fallow rice (FR). Within each year, FNRE values were not significantly different across nitrogen (N) rates and treatments.

		FNRE (%)	
Year	N rate (kg N ha^{-1})	CR	FR
2021	115	76.8	70.5
	155	69.2	67.0
	185	70.0	70.8
	225	66.7	63.6
	260	70.7	62.7
2022	115	66.1	58.9
	155	52.7	60.1
	185	59.6	66.0
	225	61.6	60.4
	260	65.0	57.2
2023	90	31.5	52.4
	120	47.9	49.6
	150	53.8	55.6
	180	55.7	42.9
	210	56.6	51.0
Average		60.3	59.2

Reflections and Learnings

Chapters end,

Papers sent.

Beyond the manuscript,

Perhaps more to comprehend.

Such was the journey,

Of tears and joy,

Failures and successes,

Arguments and collaborations.

So here I took some time,

To think beyond the science,

And reflect on the self, people, and humanity behind the degree.

At the important junctures of life, reflections go a long way. Life moves fast, and taking time for oneself to rationalize a journey is often needed, but forgotten, in our busy world. Writing is a way for me to look back at the processes that got me here, understand myself better, and chart a path forward. Beyond the scientific findings, many lessons have shaped me as an individual. So the heart opened, words spilled and ink flowed - for me, and those who want to hear my story.

Working hard, working smart, and trusting the process

All considered, I have done a good job with great efficiency. From the start of my undergrad to the end of graduate school at UC Davis, it took five and a half years – two for the BSc and three and a half for the PhD. During this time, I got four peer-reviewed articles published – three as first author, and one as second author. My grades were exceptional and I maintained a 4.0 throughout my time at Davis. I have active research projects going on and am headed into an excellent postdoctoral position at Stanford. Perhaps there are more achievements to list, but these tangible metrics are a representation of working hard and working smart. To you Zhang, congratulations and well done.

Being hardworking is a prerequisite for success, and it is so obvious that no one needs a reminder to work hard. When starting graduate school, I was a full-on workaholic. As a novice then, I had no idea how to do field research, and learning took a lot of time and effort. I took the initiative to write logistical lists, set up workflows, and annual research plans – with constant consultations with Kapitan, colleagues, and Bruce. I was new to operating the gas chromatograph (GC) and started lab work with a malfunctioning instrument. I took the initiative to learn the principles of chromatography and spent countless hours with Martin Smith (Engineer of Shimadzu Scientific) on the phone and even more time doing surgery on the instrument – with regular decommissions sprinkled throughout the years. The same went for all other workflows – sample collection, fieldwork, lab work, and all other tasks related to research. I recall 3 am check-ins on the GC, hot and long days in the field, and quite simply the days of working like a madman. Regularly, I clocked 60 to 70-hour weeks in the first two years of graduate school. Without the hard work then, there would not be the me now. Remember always, that nothing replaces the grind when push comes to shove. 自强不息 – roughly translates to endless perseverance!

While working hard is the basis for success, working smart is also an irreplaceable part of the recipe – especially asking the right questions, and working on relevant answers. During this degree, having clear objectives and hypotheses were crucial to efficiency – especially in agronomy where each experimental cycle was a year long and there was a high price to pay for failed experiments. At the start, I sat down and reasoned what I was measuring and why I was making those measurements. While I was not completely clear how the collected data could be translated into manuscripts, this process allowed me to ensure that the data collected could enable me to answer the research question. Consequently, the vast majority of the data collected were relevant and were used for publication. I had always wondered why my mentors repeatedly asked me, “what is the hypothesis”. This question was not just an intellectual exercise, but rather a means for us to understand if we are measuring the right things to answer the questions that we asked – without which I might have “gone fishing”. While asking the right questions was important, asking them to the right people was equally important. For statistics, it became much easier when I asked for feedback from individuals who worked in R, specifically with colleagues in the Pittelkow group and the DataLab. When analyzing POC and MAOC as an explanation for methane cycling patterns, working with the Gaudin group who already have workflows and personnel actively doing the research was monumentally easier than attempting to set up my own protocols. And so, moving forward, a good working definition of working smart may be to ask the right questions, and tap on those that can most directly help you in answering them.

Despite a good run, I often felt a sense of inadequacy. In part, this was driven by the state of academia and the sheer level of competition one has to face in this career. "Publish or perish" was taught to us early, and being consistently productive was an expectation that was difficult to achieve. Comparing myself to my predecessors and the profiles of current faculty applicants, my body of work is small, to say the least. The pressure to publish resulted in a non-optimal mindset for improvement and at times led to abrasions with the authorship team. I remember that chapters one and two both took close to ten correspondences before we were comfortable submitting the work. When I got feedback from mentors, I sometimes handled it inappropriately. Being a results-driven individual, I aimed to get my work published in the most efficient way possible. I felt disheartened because each round of revisions seemed like another barrier. However, each revision helped to eliminate errors and better crystallize the ideas that we were trying to express. Of course, this does not suggest endless edits – and I have worked to better communicate with co-authors and mentors as the work grows, especially in the last year of this degree, reducing instances of friction. Towards the end, I took comments and feedback as learning opportunities to better my work. This shift in mindset enabled greater receptiveness and motivation to keep the editing going.

I have come to realize that there is no need to rush and that time itself is often the essence. One really has to learn how to trust the process. The work will look right when enough work has been put in and the time is ripe. More importantly, building a body of work takes time – a lot of time – and each journey is unique, and possibly less linear than perceived by bystanders. Keep working hard and smart. And when due diligence has been performed, be gentle on yourself and trust in the process.

Communication – spoken and written

I spoke; they heard.

In a room of people,

A tension just right.

I read; they saw.

In a sea of literature,

Words and images that stuck.

So I wondered.

What made me hear more of what you said?

What made me read more of what you wrote?

Tones bright, visuals a sight, text that championed great insight.

Clarity and brevity,

The depth just right.

An ease that felt so natural,

Your world melted into mine.

Having a conversation my dear, is an act of love.

Attempting to make any content more accessible or easy to understand is a laborious task. On the individual level, one has to understand the content thoroughly - that in itself is challenging. Communicating that understanding and synthesis is even more difficult. Doing it well and having a conversation with the world, is stunning and truly an act of love.

Throughout graduate school, I have listened to a good number of talks. Of those, there was a sizeable portion where I had absolutely no clue what was going on. I reasoned that this was due to a lack of technical knowledge on my part. However, my perspective truly shifted when I attended the New Frontiers conference at Corteva in 2024. The lineup of speakers were some of the best scientists in agriculture. It was incredible, to see with my own eyes and hear with my own ears, the ability to communicate research effectively and eloquently. A testament to their craft was me feeling refreshed at the end of the day – juxtaposed against the fatigue I felt at other conferences. They articulated their scientific questions well, shared sufficient details about methodologies such that it provided appreciation but not overwhelm, and also presented findings succinctly such that I could remember a takehome message. The difference when one made an effort to make it easier for an audience was immaculate.

For myself, I have also developed my own style - less is more, slow is clear, practise and practice. Often, the task is to communicate complex ideas in a short amount of time. For the presenters, there is no time to say everything - so we need to say what matters. For the listeners, there is limited bandwidth. Realistically, if the majority of an audience can remember two to three take-home messages after a talk, that was probably a good run.

To that end, I have attempted to be clear and succinct in my presentations. My slides had few words. I spoke slowly without rushing, giving myself sufficient time to deliver my pitch, and allowing the audience to sonically receive what I was saying. When suitable, I leveraged visual animations and figures to explain concepts. For each of my presentations, I practiced a lot. For the talk at the International Temperate Rice Conference (12 min), I did 4 full rehearsals, New Frontiers (8 min), eight, and my exit seminar (50 min), five. As a theatrical artist and instructor, I have always touted that nothing replaces rehearsals and practice – and I am glad that I walk the talk. These strategies appear to be working out. For one, people are asking questions – suggesting some level of understanding and engagement, if not at least interest. Members of the audience have also approached me and shared that the material was easy to understand and have sent their congratulations. While there is not formal evaluation, these are good signs of a functional recipe. I am gratified that people do appreciate it when one takes an effort to have a conversation.

Beyond the spoken, there was the written. Cameron pitched that before any writing takes place, it is strategic to have in place all the figures and tables. Bruce pitched that in a given piece of work, three to four key ideas that the work hopes to convey need to be clearly outlined and visible throughout the text. At this stage, these have become my standard workflows because they work. Having the main messages with the results answers a rather basic question – is what I am hoping say valid? With a main framework in place, it becomes much easier to articulate findings tightly and concisely. Many of the lessons of the spoken form are equally relevant here. Practise, is often needed. First drafts are usually imperfect and that is normal. Re-drafting helps to correct mistakes and refine ideas. And as one does this more and more, it builds aptitude. Repetition is a boring process, but there are no shortcuts.

Interestingly, the success of the “written” is sometimes contingent on the visual. It is becoming increasingly common for readers to read the title, abstract, and key points, look at a couple of figures, head on for the conclusion, and call it a day. When attending a genetics class (GGG201A), the instructor noted that his favorite type of paper was when figures and tables could tell the story of the research on their own – what was done and what are the main findings. And so in my work, I have also looked to visual storytelling as a strategy. Can a reader scan through the abstract, figures and tables and understand my research? I strive to have “yes” as an answer. It may be unfortunate that we are living in times where the likelihood of someone reading a paper from top to bottom is low. Yet, when being flooded by a sea of literature and information, the trend of strategic skimming will probably be here to stay. So for the written, perhaps both the words and the pictures are equally important in helping the work reach a wider audience.

To have a fruitful conversation, there is a process of giving, receiving, processing, and engaging with information. No one owes me their time and attention, and thus the onus is on the speaker and writer to make their case well. The baseline here is to do the work such that the audience need not work harder than they have to.

Openness in science and a higher purpose

Moving forward as a scientist, I hope to do high-impact work and ask novel questions. From a pragmatic standpoint, the problems and questions that society and scientists are hoping to solve are multi-dimensional and require an interdisciplinary approach, necessitating a great degree of openness and collaboration. I have been, and foresee myself, to be involved in interdisciplinary collaborations. As written above, to have a conversation, is an act of love. Being able to collaborate and communicate in interdisciplinary research is also a love language. During graduate school, I have invested a sizeable portion of my time to acquire knowledge outside of my immediate expertise. Specifically, these included weed science and herbicide physiology, plant breeding, and genomics. Although these subject areas did not contribute directly to this dissertation, they have allowed me to appreciate a greater diversity of research in the field and opened doors for collaboration. While learning and expanding knowledge is useful, I also need to develop my niche. I need to be a specialist in my space while being able to communicate with those adjacent to my field. Interacting with individuals working in tremendously different systems has often challenged the pedagogies I hold and have at the very least, been an attempt to more fully answer the questions I have asked – in terms of the plant and the soil bio-geo-chemistry.

The completion of a PhD is the end of a chapter and the start of countless others. The journey was a tough one, and I am grateful to be on the receiving end of guidance and mentorship. While the next couple of years will be a mission to publish as much as humanly possible, I also hope that I can contribute to the experiences of others and pay it forward. A faculty I met once told me, “If one only wanted to do research, then one would only hire technicians and not graduate students. But it is not all about productivity, we also have the responsibility to train the next generation of scientists." I am still very much learning and getting trained as a scientist in the postdoc stint. However, this is also a position where there are expectations to provide guidance or input to others, and I hope to contribute to the learning experiences of others. As I progress in my career, I hope that amidst all the hustles and craziness (especially if I stay in academia), I will remember that "training the next generation" is a responsibility we all share. Importantly, this journey has taught me a great deal of patience and empathy. From where I am from, there was very little appetite and room for incompetence. Mess up, and one is out. I have witnessed firsthand those around me start from zero and transform into excellent researchers. Patience is the essence – because learning takes time and everyone learns at a different pace, and empathy is of importance – because understanding what one needs to improve and providing the right inputs is key for myself and others to move forward.

As I transition from a department of "plant sciences" to "earth system science", I truly wonder what I will call myself in two years. For now, I am happy to call myself an agronomist – but as the new generation, I do intend to challenge what that title may entail and blaze my own trail. There are mouths to feed and it is getting monumentally challenging to grow food on Earth. To you Zhang, ask the bold questions and do the rigorous science – wherever you may be. Continue to dream of feeding the world, with all the humanity in your heart.

So here was a checkpoint,

That one worked so hard for.

But beyond here,

Lies a scene I never once saw,

And many many winding roads.

Let’s take a trip I say,

And dive into the unknown,

The edge of knowledge,

With a spirit strong, an open mind, and a heart of loving kindness,

In search of a better world,

And to keep the promise I once made.