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

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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 carbon (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.

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,

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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 and 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 and 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 and 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 and 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, 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. This will shed light on the cross component effect for mitigation practices to be effective in one component but 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).

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 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).

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).

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 unit (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 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 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.

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 four components of net system emissions.

Geographic specificity	
Southeast Asia/SEA, Malaysia, Vietnam/Viet Nam, Indonesia, Myanmar/Burma, Singapore, Brunei/Brunei Darussalam, Cambodia, Philippines, Laos/Lao PDR, Thailand	
Components	**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

Table 2
Summary table of literature search. "Number of duplicates", "Records Screened", and "Records Excluded" are totals and not sorted by components. 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 studies shortlisted, refer to supplementary material. GHG stands for greenhouse gas and SOC stands for soil organic carbon.

	Field GHG emissions	Energy inputs	Residue utilization beyond the field	SOC change	Total
Hits	564	779	522	108	1973
Number of duplicates					467
Records Screened					1506
Records Excluded					1407
Records Retained	38	33	18	12	99

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 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

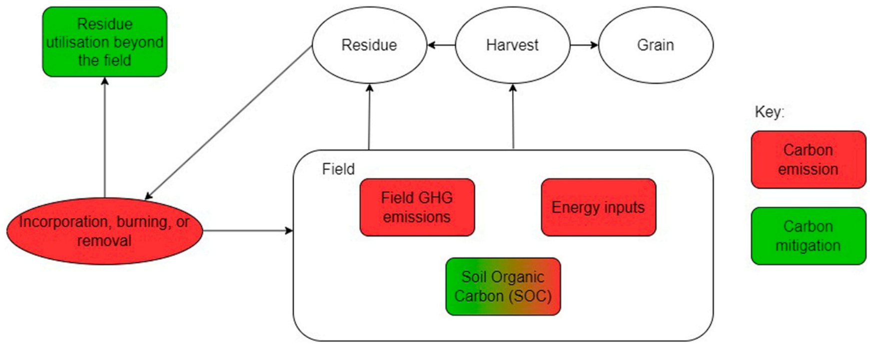

Fig. 1. Schematic of net system emissions conceptual framework guiding the literature search and review (each colored box represents a pool of C flux, with red representing emissions and green representing mitigation).

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 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.

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 and 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% and 47.1% (mean 23.3%) and 6.1–63.4% (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 80.3–96.0% (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 40.2–37.8% (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

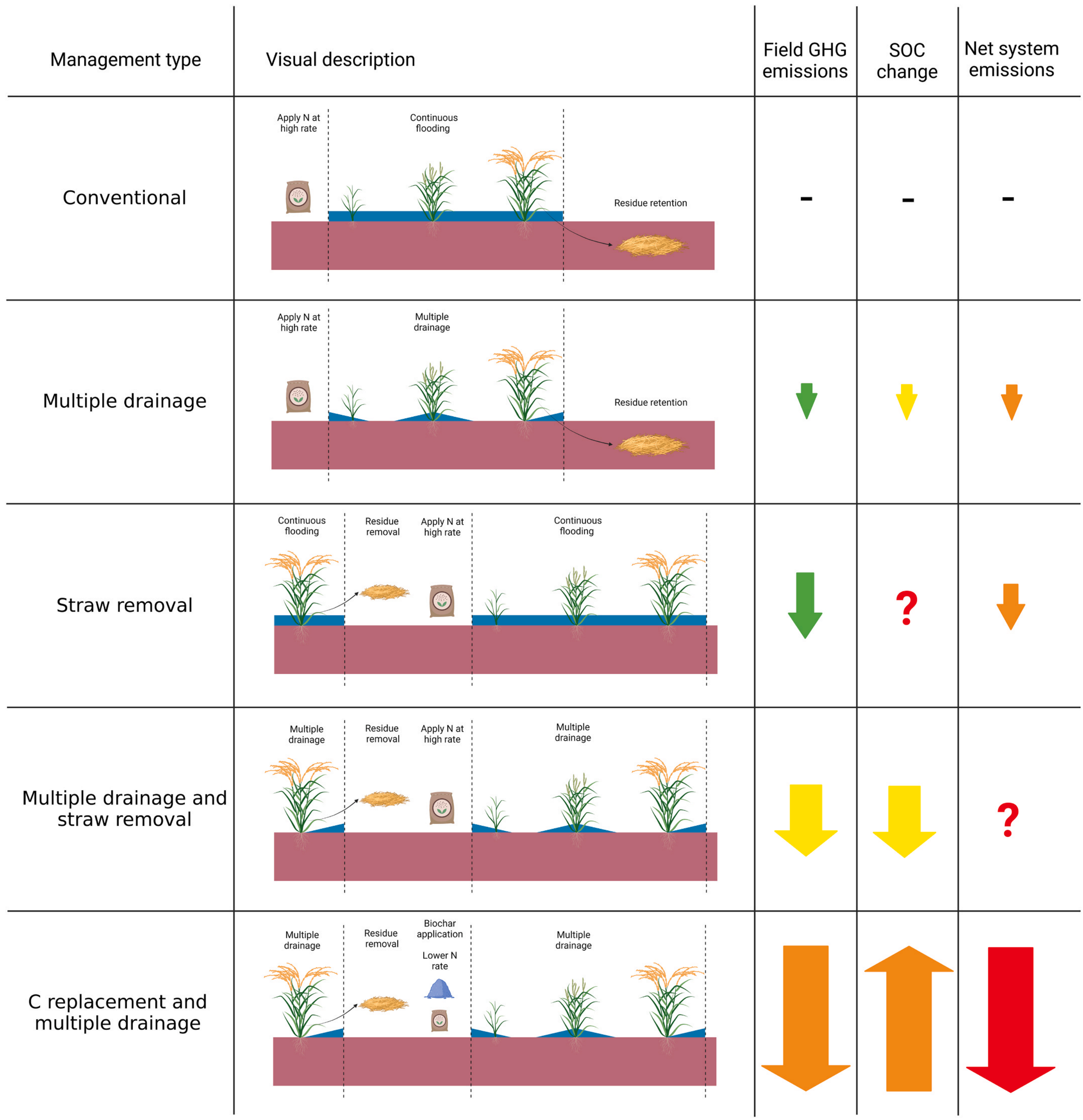

Fig. 2. Conceptual figure summarizing the relative impact of four mitigation strategies (multiple drainage, straw removal, multiple drainage and straw removal, and C replacement and multiple drainage) on the components of field GHG emissions, SOC change, and net system emissions compared to a conventional 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. This figure was created with BioRender.com.

dependent on the manufacturing process and feedstock.

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).

Table 3
Mitigation potential of technical options aimed at reducing GHG field emissions for additional studies not found in Yagi et al. (2020). Wet (WS) and dry (DS) season 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		Continuous flooding with and without straw retention	Samoy-Pascual et al., (2019)
	Vietnam	5245	17030	5838	23540	Continuous flooding	Tirol-Padre et al., (2018)
	Indonesia	4843	13342	6413	17861	Continuous flooding	Tirol-Padre et al., (2018)
	Thailand	86	746	-	1190	Continuous flooding	Tirol-Padre et al., (2018)
	Philippines	325	2853	-1587	11333	Continuous flooding	Tirol-Padre et al., (2018)
Fallow drying	Philippines	2660–3181	3314	59–343	483	Continuous flooding. 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)

DS, dry season; WS, wet season.

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 emissions of energy usage pinpointed agrochemicals, especially synthetic N fertilizers (Bautista and 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 and 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 and 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 and 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 18 studies assessed options for residue utilization beyond the field to produce energy or substitute materials used in other agricultural production systems. Nine of these quantified emissions mitigation through straw removal and subsequent electricity generation, bio-DME (dimethyl ether) production, mushroom cultivation, or bio-ethanol 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, two 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

Table 4
Mitigation potential of utilizing residue beyond the field. The four main options included electricity generation, manufacturing of bio-DME (dimethyl ether), 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)

(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.

The nine 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 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 and 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 10 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 1980–1990–1990–2010 (Minasny et al., 2012). Arunrat & Pumijumnong (2017) found a similar range of SOC increase over 10 years in Thailand.

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 3.74–26.74 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 and Lal, 2013).

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

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

		SOC change (Mg C ha^{-1} $year^{-1}$)			
Option	Country	Mean	Range	Duration (years)	Reference
Long-term data	Indonesia	0.32	-	20	Minasny et al., (2012)
	Thailand	0.44	–0.30-1.02	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 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	0.31–1.38	1	Vityakon 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)

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 was 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 can potentially promote SOC accumulation (Maneepitak and 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 treatments or practices 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.

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 6334 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

Table 6
Relative contribution of the 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 kg CO_2 eq ha^{-1} yr^{-1} which includes one dry season and one wet season. SOC change is difficult to estimate and has high uncertainty due to a lack of long-term data from empirical studies.

	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

value has high uncertainty.

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. 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.

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 of one to two drainage events to 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; 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 0.7–1.3 t 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 (Wassmann, 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.

Declaration of Competing Interest

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

Data availability

Calculations and other relevant data related information can be found in supplemenatry materials.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2023.108812.

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