Greenhouse gas emissions and mitigation in rice agriculture

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Abstract

Rice paddies supply half the global population with staple food, but also account for ~48% of greenhouse gas (GHG) emissions from croplands. In this Review, we outline the characteristics of GHG emissions (CH_4 and N_2O) from paddy soils, focusing on climate change effects and mitigation strategies. Global mean annual area-scaled and yield-scaled GHG emissions are ~7870 kg CO_2e ha^{-1} and 0.9 kg CO_2e kg^{-1}, respectively, with 94% from CH_4. However, emissions vary markedly, primarily reflecting the impact of management practices. In particular, organic matter additions and continuous flooding of paddies both stimulate CH_4 emissions, whereas fertilizer N application rate is the most important driver of N_2O emissions. Although contemporary changes in emissions are uncertain, future elevated [CO_2] and warming are projected to increase CH_4 emissions by 4-40% and 15-23%. Yet integrated agronomic management strategies—including cultivar, organic matter, water, tillage and nitrogen management—offer GHG mitigation potential. In particular, new rice variety, non-continuous flooding, and straw removal strategies reduce GHG emissions by 24%, 44%, and 46% on average, respectively. However, approaches need to be optimized based on seasonal CH_4 emission patterns, necessitating improved quantification and reduced uncertainty in regional and global GHG estimates, especially in low latitudes.

[H1] Introduction

Rice is a vital crop for food security and human nutrition, with global rice paddies currently occupying ~1.7 million km^2 (ref [1]). It is the main staple food for more than half of the global population and provides 20% of dietary energy supply[2]. While China and India dominate consumption, global consumption has increased markedly, growing from 157 million tons in 1960, to 520 million tons in 2022 (ref [3]) . Consumption is further expected to rise by an additional ~6% up to 2030 (ref [4]). China, India, Bangladesh, Indonesia and Vietnam contribute the most to rice production, which in 2020, totaled ~757 million tons[5].

Although an important food crop, rice paddies are a major source of greenhouse gas (GHG) emissions. For example, rice contributes 22% and ~11% of total agricultural methane[6] (CH_4) and nitrous oxide[7] (N_2O) emissions, respectively. Owing to these high CH_4 emissions, rice has the highest area-scaled (that is, GHG emissions per unit land area) and yield-scaled (that is, GHG emissions per unit yield) emissions of all food crops[8,9], and despite increasing soil organic carbon (SOC) stocks[10–12], are generally considered a net source of GHG[1].

In addition to soil properties and agricultural management practices, these GHG emissions are strongly influenced by climatic conditions, including temperature and precipitation[13,14] (**Box 1**). For instance, warming can increase substrate availability for methanogens as well as their abundance, resulting in higher CH_4 emissions[15–17]. Indeed, meta-analyses indicate that warming enhances CH_4 emissions from rice paddies by 15-23% (ref [18–20]). Moreover, warming can increase N_2O emissions by increasing the abundance of nitrite reductase genes[21].

This combination of climatic change and expansion of rice growing area have, therefore, contributed to rising anthropogenic GHG emissions. Estimates of current annual CH_4 and N_2O emissions from rice paddies total 24-31 Tg yr^{-1} (refs [5, 6]) and 130 Gg yr^{-1} (ref [14]), respectively. Accordingly, there is a need for mitigation. Agricultural practices offer such possibility because they alter soil C, N, and O_2 availability. For instance, high-yielding rice cultivars can reduce CH_4 emissions owing to higher root O_2 release[22], mid-season drainage can reduce CH_4 emissions by 52% (ref [23]), and returning straw during the non-rice growing season could potentially reduce global CH_4

emissions by 5.4 Tg yr^{-1} (ref [24]).

In this Review, we outline GHG emissions and mitigation potential of rice agriculture. We begin by evaluating GHG emissions from rice paddies across spatial scales, before then discussing the impacts of climate change on these emissions; focus is placed on empirical, field scale research (**Supplementary text**). We follow with a discussion of mitigation strategies that can reduce GHG emissions from paddies, offering insight on which are most likely to be successful. Finally, we identify research gaps that need to be addressed to achieve further GHG reductions.

[H1] Spatiotemporal characteristics of GHG emissions

GHG emission rates vary depending on climatic conditions and agricultural practices. The spatiotemporal characteristics, overarching dynamics, and longer-term changes of in-situ GHG emissions (**Supplementary Methods; Supplementary Data**) are now discussed.

[H2] Geographic characteristics

Averaging across all in-situ available observations provides a reasonable indicator of overarching rice paddy GHG emissions and their characteristics. Annual CH_4 and N_2O emissions are 283 kg ha^{-1} and 1.7 kg ha^{-1}, respectively (**Supplementary Table 1**). These values can be combined and converted to global warming potential (GWP) over a 100-year time horizon[6] to give area-scaled and yield-scaled GHG emissions. Area-scaled emissions are estimated at 7870 kg CO_2e ha^{-1} yr^{-1}, with CH_4 contributing ~94% of that total, and yield-scaled emissions at 0.9 kg CO_2e kg^{-1}.

However, there is substantial heterogeneity in annual GHG emissions (**Fig. 1**), varying up to two orders of magnitude across paddies, even within the same country (**Supplementary Data 1**). Embedded within this heterogeneity is latitudinal dependence. For instance, mean CH_4 emissions reach 614 kg ha^{-1} yr^{-1} from 10ºS-10ºN, generally declining to 272 kg ha^{-1} yr^{-1} at >40ºN (**Supplementary Table 2**). N_2O emissions, in contrast, exhibit less latitudinal dependence, peaking at 1.95 kg ha^{-1} yr^{-1} for 10-20ºN and dropping to a more consistent 1.29 kg ha^{-1} yr^{-1} at 10ºS-10ºN (**Supplementary Table 2**). Given the dominance of CH_4 in quantifying area-scaled and yield-scaled GHG emissions, these also peak at 10ºS and 10ºN, totaling 18,171 kg CO_2e ha^{-1} yr^{-1} and 1.19 kg CO_2e kg^{-1}, respectively (**Supplementary Table 2**). These values should be interpreted with caution given the low number of in situ observations in low-latitude regions.

At the country level, China, India, and Indonesia have the largest rice area and so are the largest countries in terms of total CH_4 emissions, contributing 22-38%, 11-19%, and 7-9% of the 24-37 Tg yr^{-1} global total (), respectively[5,25]. However, the Philippines has the highest area-scaled and yield-scaled CH_4 emissions, reflecting the impact of high temperatures and the fact that most rice paddies are continuously flooded. In contrast, China and Bangladesh have the lowest yield-scaled CH_4 emissions mainly owing to high rice yield and widely applied non-continuous flooding (NCF) practices[9].China, India, and Indonesia also have the largest total N_2O emissions from rice cultivation, contributing 27%, 17%, and 15% of the global total (130 Gg yr^{-1}), respectively[14].

[H2] GHG emission drivers

Management practices have a key influence on GHG emissions from rice paddies, overshadowing the impact of climate and soil type. For CH_4, organic matter (straw and manure) and water management are the most important predictors of emissions (**Supplementary Fig. 1a**). Organic amendments are commonly applied to increase soil fertility, crop yields, and soil C sequestration[26–28]. However, the added organic matter provides substrate for methanogens, thereby stimulating CH_4 production. Higher CH_4 availability, in turn, provides substrate for methanotrophs which consume CH_4. Production generally outweighs consumption[29], resulting in a net increase in CH_4 emissions, but the response varies with the type of organic matter that is being added. For instance, livestock manure tends to increase CH_4 emissions by 60% on average, whereas straw application increases CH_4 emissions by an average of 92% (ref [30]). Moreover, long-term organic matter addition can shift the community composition of methanogens and methanotrophs, altering the relative abundance of acetoclastic and hydrogenotrophic methanogens[31–33] and increasing the abundance of methanotrophs with preference for high CH_4 concentrations[34]. Thus, CH_4 emissions with straw addition could decrease over time, owing to increased abundance of methanotrophs, presumably by stimulating root growth and O_2 release into the soil[29,34].

Water management is also a key control of CH_4 emissions. Compared to continuous flooding, NCF practices (including mid-season drainage, intermittent irrigation, alternate wetting and drying)[35] typically reduce the abundance and activity of methanogens (**Supplementary text**). These practices also increase soil O_2 concentrations, soil *Eh* (redox potential), and the abundance and activity of methanotrophs. Together, these aspects lower CH_4 emissions[36–38] by an average of 53% during the rice season[23]. Extrapolated to the global scale, multiple drainages during the rice season could potentially reduce CH_4 emissions by 4.1 Tg yr^{-1} (ref [24]). In addition, NCF can exert carry-over impact on GHG emissions in following fallow and cropping seasons, which has not been taken into consideration in previous assessments. For instance, mid-season drainage reduced CH_4 emissions by ca. 60% during the fallow season when the paddies were permanently flooded[39].

Different factors control N_2O emissions. N application rate is considered the most important driver (**Supplementary Fig. 1b**), increasing N_2O emissions by 182% on average compared to a situation without N fertilizer addition[40]. The underlying mechanisms are likely attributed to increases in substrates for both nitrification and denitrification, and decreasing soil pH[41–43]. The global direct N_2O-N emission factor (the percentage of applied N emitted directly as N_2O-N) for rice is estimated at 0.52% (0.15% to 1.3%)[14], roughly consistent with 0.3% in continuous flooding systems and 0.5% in NCF systems suggested by IPCC Guidelines[44]; higher values in NCF systems reflect that nitrification and denitrification reactions are strongly reduced under continuously flooded conditions.

For area-scaled and yield-scaled GHG emissions, organic matter management is the most important driver (**Supplementary Figs. 1c&d**). Indeed, organic matter addition increases area-scaled total GHG emissions by 66-85% and yield-scaled emissions by 37-87%[30,45], emphasizing the importance of optimizing organic matter management to lower GHG emissions while maintaining high rice yields.

[H2] GHG emission dynamics

Rice GHG emissions also exhibit variability over various temporal scales, including the growing season. These growing season emission dynamics are largely determined by management practices and environmental conditions, and for CH_4, can be divided into 4 typical temporal patterns (**Fig. 2; Supplementary Table 3**); there are no generally occurring patterns for N_2O, though peaks can be found after N fertilization events or during draining periods[46–48].

The first of these CH_4 patterns is a single, early emission peak (**Fig. 2a**). In this pattern, CH_4 fluxes increase after flooding for rice transplanting or sowing because straw and/or stubble from the previous season provides substrate for CH_4 production. CH_4 emissions peak at the tillering stage, and in most cases (especially with midseason drainage and intermittent irrigation) decrease sharply to near-zero until harvest[46,49,50]. About 40% of the observations follow this temporal pattern, particularly in China (n=168, 52% of Chinese observations), Korea (n=14, 35%), and Vietnam (n=16, 37%) (**Supplementary Table 3**).

The second pattern exhibits two emission peaks (**Fig. 2b**). Here, CH_4 fluxes increase after transplanting or sowing, peak at the tillering stage and then decline. Emissions typically exhibit a 2nd peak at booting or heading stage, when reflooding and rice roots provide substrates for CH_4 production. CH_4 production subsequently decreases until harvest[33,49,51]. About 27% of total observations fall into this category, including in China (n=93, 29%), the Philippines (n=9, 29%), and Vietnam (n=12, 28%).

The third pattern consists of a single, late emission peak (**Fig. 2c**). In this case, initial CH_4 fluxes after transplanting or sowing are low, because low temperatures slow down CH_4 production. Other factors that can cause low initial CH_4 fluxes are decomposition of organic matter from preceding crops under aerobic conditions during the preceding fallow season (thereby reducing substrate for CH_4 production), and aerobic soil conditions of dry-direct seeded rice. As the growing season progresses, CH_4 fluxes gradually increase owing to rising temperatures and increasing root exudates. CH_4 then peaks at the booting or heading stage owing to root exudates. The decrease in emissions after heading arises because of pre-harvest drainage and low temperature[52–54]. About 24% of total observations fall into this classification, particularly in the USA (n=14, 61%), Japan (n=24, 58%), and India (n=18, 43%).

The final pattern is characterised by near-continuous emissions that resemble a bell-shape (**Fig. 2d**). Here, CH_4 fluxes increase after transplanting or sowing, remain high owing to continuous flooding and high temperature, and then decrease until draining at the end of rice season[55,56]. About 8% of total observations follow this pattern, most often in the Philippines (n=9, 29%) and India (n=8, 19%).

In addition to emissions during the growing season, as characterised by the 4 temporal patterns, rice fields can also emit CH_4 during the fallow season. These fallow season emissions are particularly common in Mediterranean, subtropical and temperate areas, where winter flooding and rice straw incorporation after harvest is a common practice[57]. Agricultural practices during the fallow season also affect GHG emissions during the following rice season. For instance, a dry fallow season reduces CH_4 emissions during the subsequent growing season in comparison to winter flooding[11,58]. However, the contribution of these fallow season emissions to total global paddy emissions is unclear owing to the lack of measurements.

[H2] Long term trends in global emissions

In addition to these seasonal characteristics, GHG emissions from rice paddies have also evolved in the longer-term owing to changing agricultural practices[59], namely water management and organic matter management. One pool of thought suggests that CH_4 emissions changed quite substantially. For instance, the IPCC sixth assessment report[6] outlines that CH_4 decreased from 45 to 29 Tg yr^{-1} between 1980-1989 and 2000-2009, before increasing to 31 Tg yr^{-1} over 2008-2017 . Similar changes are derived from the EDGAR v7 dataset[25]. However, other datasets suggest greater stability in global CH_4 emissions. FAO data[5], for instance, indicate emissions varied from 22 Tg yr^{-1} in 1980-1989, 23 Tg yr^{-1} during 2000-2009, and 24 Tg yr^{-1} during 2010-2019.

Steady increases in average rice yield per hectare (**Supplementary Fig. 2**) have reduced global yield-scaled CH_4 emissions by 38-55% from the 1980s to 2010s (ref [5,6,25]), as observed in most countries (**Supplementary Fig. 3**).

Although corresponding data are not available for N_2O emissions from rice paddies, the increase in chemical N fertilizer use and N surpluses, combined with the rising popularity of NCF practices during the same period[5,9,60], suggests that global N_2O emissions from rice paddies have also increased[61].

[H1] Climate change effects

By the end of the 21st century, atmospheric CO_2 concentrations are predicted to be almost 1000 ppm and average global surface temperature to have risen by 1.4-4.4°C[6]. Elevated atmospheric CO_2 concentrations (eCO_2), warming and other climate change impacts will have substantial effects on GHG emissions from rice paddies, as now discussed.

[H2] Elevated CO_2 concentrations

eCO_2 is thought to increase CH_4 emissions from rice paddies. Indeed, under mean +180ppm conditions, CH_4 emissions are typically enhanced by 20-40% (ref [62,63]), although with marked variability (**Fig. 3a**). These enhancements occur through increases in leaf photosynthesis and rice growth[64,65], which, in turn, increase available organic substrate for methanogens (rhizodeposits, root exudates and litter) and subsequently methanogen abundance[15,66]. Increased substrate availability can also affect the composition of soil microbial community, for example by causing a shift from acetoclastic (using acetate as an electron acceptor) to hydrogenotrophic methanogens, and decreasing the relative abundance of methanotrophs with preference for high O_2 and tolerance to low CH_4 concentrations[15,16].

These eCO_2 effects on CH_4 emissions could wane over time, as evidenced by two long-term FACE experiments and a pot experiment[67,68] . This reduction occurs as high root O_2 release and low NH_4^+ concentrations increase the abundance of methanotrophs[67,68]. If this pattern is representative for real-world cropping systems, then previous short-term experiments might have overestimated the effect of rising CO_2 concentrations on future CH_4 emissions.

Agricultural practices interact with the effects of eCO_2 on CH_4. For instance, FACE experiments indicate that eCO_2 and N fertilization collectively enhance CH_4 emissions compared to eCO_2 alone [69]. This enhanced effect possibly arises because the increase in soil N availability reduces the C/N ratio of the plant residues, promoting plant residue decomposition, and increasing root growth and rhizodeposition[70,71]. Water management similarly has an impact; under eCO_2 and continuously flooded conditions, CH_4 emissions increased by 50%, whereas there was no effect under NCF conditions. This effect is explained by O_2 availability during frequent drainage that restricts the growth of methanogens, even under eCO_2 (ref [72]).

Straw management also modulates the effect of eCO_2 on CH_4 emissions, explaining more variability in the response of CH_4 emissions to eCO_2 than a wide range of environmental and experimental factors (water management, N fertilization, experimental duration)[73]. Indeed, a mesocosm experiment indicates that eCO_2 had no effect on CH_4 emissions from paddy soils with straw incorporation because rhizodeposition was not key substrate for methanogens[73]. Overall, accounting for the interactions between CO_2 and straw management and the current coverage of straw incorporation, eCO_2 is estimated to enhance global CH_4 emissions from rice agriculture by only 3.7%, much lower than suggested in earlier experiments without straw incorporation[62,63].

eCO_2 also has an impact on N_2O emissions from rice paddies. While, on average, eCO_2 elevates N_2O emissions by 27%, there is large variation in treatment effects (**Fig. 3b**)[74]. This variation might largely be caused by differences in experimental duration[74]. Short-term eCO_2 often stimulates N_2O emissions by increasing the availability of labile soil C and NO_3^-, thus promoting denitrification[63,75]. Yet, long-term results suggest that eCO_2 reduces N_2O emissions, mainly by a reduction of soil N availability[68,76]. Given the impacts on CH_4 and N_2O, eCO_2 also increases area-scaled GHG emissions by 16% on average[63]. However, the effects on yield-scale GHG emissions are inconsistent; some field experiments report increases of ~39%[77] while others show increases of ~3%[75].

[H2] Warming

Like eCO_2, temperature changes can also influence GHG emissions from rice paddies[18]. On average, experimental warming (with a range of temperature increases) enhanced CH_4 emissions by 15-23%[18–20] (**Fig. 3a**). Indeed, 1 °C warming increased CH_4 emissions from China's paddies by 12.6% (ref [19]).

These increases likely occur by enhancing substrate availability for methanogens, the ratio of CH_4 to CO_2, and methanogenic activity[15–17]; and reducing soil redox potential to favor CH_4 production through decreased O_2 solubility in water/soil solution, accelerating the consumption of O_2 and other electron acceptors by microbes[78]. However, these average increases in CH_4 emissions with warming mask substantial variability in experimental results (**Fig. 3a**), likely reflecting dependence on background temperature range[19]. For example, while CH_4 emissions increased by 25.6% from rice paddies at medium temperature range (23°C-30°C), no effect is observed at low (<23°C) or high temperatures (>30°C)[19]. This sensitivity arises through stimulation of methanogens within the biologically favorable medium temperature range[19,79], although this hypothesis does not fully explain the stagnant response to warming at low background temperatures. Given that the mean temperature of the growing season in most rice growing regions falls between 23-30°C, anthropogenic warming could thus enhance CH_4 emissions from rice paddies.

Warming also stimulates N_2O emissions from rice paddies by 26% (ref [20]; **Fig.3b**). These enhanced N_2O emissions likely occur by accelerating soil organic matter, increasing the inorganic N availability for N_2O production[18,80,81]. Moreover, warming could stimulate N_2O emissions by affecting the soil microbial community, increasing the abundance of the N_2O reductase, ammonia-oxidizing and nitrite reductase genes in archaea and bacteria[21]. Although there are numerous reports on the effects of warming on either CH_4 emissions or N_2O emissions, quantification of GHG species changes combined—and therefore on area-scale and yield-scaled emissions—is limited[75,77].

[H2] Combined elevated CO_2 and warming

Given that changes in eCO_2 and warming are concurrent, it is important to investigate the combined effect of these environmental drivers (**Fig. 3**). While determination of these combined effects is limited, there are synergistic effects -- mean effects are higher than those of only eCO_2 or only warming. For instance, the combined effects of experimental warming and eCO_2 enhance CH_4 and N_2O emissions by 71%and 36% on average, respectively, compared to ambient temperature and ambient CO_2 levels (**Fig. 3; Supplementary Table 4**). Accordingly, the effects of anthropogenic climate change on GHG emissions from rice paddies might be higher than currently assumed from single-factor experiments.

[H2] Other climate change impacts

Beyond eCO_2 and warming, other anthropogenic perturbations to the climate system are also anticipated to influence rice GHG emissions through impacts on rice plant growth and soil microbial activity[82–85]. For instance, sea level rise –predicted to rise by up to 0.28-0.55m by the end of this century even under the very low GHG emissions scenario) will cause economic and environmental problems such as soil salinity, loss of agricultural land area and yield declines[86–88], with lowland rice fields being particularly vulnerable. Increases in soil salinity tend to reduce CH_4 emissions[50]. Moreover, enhanced frequency and severity of extreme weather events (including heat, drought, heavy precipitation, and compound events) will strongly affect rice cropping systems[82,89]. For instance, extreme temperatures are estimated to reduce global rice yields by 33.6% in the 2090s[28]. However, the effects of sea level rise and extreme weather events are difficult to mimic under field conditions, and so their effect on GHG emissions remains highly uncertain.

[H1] Mitigation strategies

Management and agricultural practices have a key role in mitigating GHG emissions from rice paddies, including through rice variety selection, water management, organic and mineral fertilization, tillage, crop establishment, and other soil amendments (**Figs. 4 and 5; Supplementary Tables 5 and 6**). These mitigation strategies are now discussed.

[H2] Rice variety selection

Rice plants regulate CH_4 emissions through two primary mechanisms[90], the balance of which determines differences in CH_4 emissions among rice varieties, highlighting mitigation potential through cultivar selection. The first of these mechanisms is that rice plants provide substrates for methanogens via rhizodeposition, accounting for 40-60% of the organic C as CH_4 substrate from the booting stage onwards[91–93]. The production of CH_4 is also affected by the quality of root exudates. For instance, roots and root exudates containing higher contents of carbohydrates increased the expression of the CH_4 production gene in the soil microbial community[94]. The second mechanism is that rice plants can stimulate CH_4 oxidation by diffusion of atmospheric O_2 via aerenchyma into the rhizosphere[90,95]. These rhizospheric CH_4 oxidation rates can reach up to 94% (ref [96]), and often increase with increasing levels of root radial O_2 release[97–99]. Thus, rice varieties with high rhizodeposition might increase CH_4 production, whereas large root systems could promote CH_4 oxidation. Nevertheless, the aerenchyma also acts as a conduit for CH_4 from the rhizosphere to the atmosphere[100].

Balancing these two mechanisms through selecting cultivars with a high harvest index (HI) or with high plant biomass offers opportunities for mitigation (**Fig. 4**). Cultivars with a high HI might reduce CH_4 emissions from rice paddies (**Fig. 4a**); a high HI increases the amount of photosynthate allocated to grains, reducing the amount allocated to rhizodeposits, which, in turn, reduces substrate availability for methanogens[101,102]. This hypothesis has been tested in two experimental approaches: reducing photosynthate allocation to grains by removing rice spikelets[101], and increasing the allocation of photosynthate to grains by a single transcription factor gene addition[102]. Both approaches reduce CH_4 emissions in continuously flooded systems by lowering rhizodeposits. However, high HI cultivars did not reduce CH_4 emissions in NCF systems[103], likely owing to high

O_2 concentrations and *Eh* limiting the growth of methanogens during the drain and reflooding stages (**Fig. 4a**)[103]. Thus, although the empirical basis is still limited, CH_4 emission mitigation through HI improvement might be limited in rice paddies when NCF practices are applied in the last half of the season.

Alternatively, CH_4 could be mitigated by selecting cultivars with high plant biomass (**Fig. 4b**); larger plants generally have larger root systems that can release more O_2,facilitating CH_4 oxidation by methanotrophs[95].. However, the relationship between rice biomass and CH_4 emissions is inconsistent[104–106], likely reflecting interactions of genotype and environments. Indeed, rice cultivars with high biomass reduce CH_4 emissions by ~24% under high levels of organic soil C, regardless of whether the C was derived from autochthonous soil organic matter or straw incorporation[22]; in this case, the abundance of methanotrophs increases more strongly than the abundance of methanogens owing to high CH_4 concentrations in soils and more root radial oxygen loss[22]. In contrast, high biomass cultivars enhance CH_4 emissions from soils with low organic C stocks, likely because organic C from root dominates the substrate for CH_4 production in these soils[22]. Yet, higher biomass might stimulate CH_4 emissions in the long term by increasing soil C input, unless the straw is removed.

Plant breeding efforts have focused on developing rice varieties with both high biomass and high yield[107–110]. Importantly, CH_4 emissions also differ among these new high-yielding rice varieties[22,106], but key traits of high-yielding and low CH_4 emissions rice varieties are unclear. In addition, rice growth duration is an important factor determining CH_4 emissions given that long-duration varieties require paddies to be flooded for longer periods of time[90]. High-yielding short-duration rice varieties might therefore have potential to reduce CH_4 emissions[111].

Rice plants can affect N_2O emissions by providing C substrates for denitrification and altering soil N availability and moisture. Observations across several rice cropping systems suggest that N_2O emissions negatively correlated with HI[112]. Reduction in HI through spikelet removal drastically increased N_2O fluxes by 67%-155% under NCF systems, because of increased root exudation and reduced plant N uptake[112]. However, the effect of rice cultivars with high biomass on N_2O emissions is still unclear.

[H2] Water management

To maximize reductions in GHG emissions with NCF practices, the number and timing of drying events and soil drying severity can be optimized (**Fig. 5**). Indeed, the effect of NCF on CH_4 emissions correlates with the total number of unflooded days, with single and multiple drying events reducing CH_4 emissions by 33% and 64% on average, respectively[23]. Other estimates suggest CH_4 emissions reductions of 29% and 45% (ref [44]), likely owing to different methodologies (statistical analysis of combined field observations vs. direct side-by-side comparison). Besides drainage duration, the severity of drainage also affects CH_4 emissions. For instance, a single mid-season drain event, if severe enough, can keep CH_4 emissions low for the rest of the season without reducing rice yields[113]. Finally, the timing of the drainage event is also important; drainage at peak CH_4 emissions maximizes CH_4 emissions mitigation from rice paddies[114].

NCF practises also influence N_2O emissions. In continuously flooded rice systems, N_2O emissions are typically low or negative throughout the growing season[46,113] given nitrification and denitrification processes during soil-drying and reflooding[115]. Compared to continuous flooding, NCF practices effectively increase soil O_2 concentrations and the activity of most nitrogenconverting microorganisms, resulting in N_2O emissions that are 105% higher, on average[23,36,38]. While in most cases N_2O emissions increase with drying events, the baseline N_2O emissions are low and these increments generally do not offset the benefit of reduced CH_4 emissions[23]. Furthermore, while NCF practices increase soil C loss and N_2O emissions, the reductions in CH_4 emissions generally outweigh these effects in terms of GWP[116]. However, exceptionally high N_2O emissions (33 kg ha^{-1} $season^{-1}$) have also been observed, leading to an overall higher GWP compared to continuous flooding[48,117,118]. This results from a field drying when soil extractable N is high; thus, the timing of the drying period and soil N need to be co-managed[48,119].

NCF practices are thus beneficial to reduce GHG emissions from rice paddies. Based on assessments of daily precipitation, water loss through crop evapotranspiration and potential percolation, such NCF practices could be implemented in 76% of global rice plant areas without reductions in rice yield[120]. However, the actual area in which NCF can be implemented will be substantially lower owing to logistical challenges of implementing paddy drainage. NCF practices are already widely adopted in China, Japan, Korea and the southern USA[35,59,121,122]. Yet, current adoption rates of NCF in some South or Southeast Asian countries are relatively low[120], probably because the rainy season presents challenges implementing NCF[123] that can be exacerbated by the low altitude of rice growing areas. In Europe, most of the rice growing area is irrigated by continuous flooding[124,125], but NCF practices have been tested in Italy[117,126] and Spain[127], suggesting there is potential to expand NCF practices in Europe. NCF practices can also be intensified further in countries with high adoption rates of NCF. For example, most rice paddies in China are continuously flooded during the early stage of the rice growing season when CH_4 emissions are very high[59]. Thus, the mitigation of further CH_4 emission through NCF is still appreciable even in systems that already apply some form of NCF during the later stages.

Water management in rice paddies therefore offers a large potential to reduce CH_4 emissions from rice agriculture[23]. To maintain high rice yields and maximize reductions in GHG emissions, draining at high CH_4 emissions stages, multiple draining events, and moderate drainage severity are recommended. During the fallow season, rice paddies should be kept unflooded, if possible.

[H2] Organic matter management

Long-term organic matter management practices (for instance, the retention or removal of straw, farm manure, green manure, cover crops, crop residue) similarly influence net GHG emissions[128,129]. Compared to straw removal, straw addition increases CH_4 emissions with a GWP that was 3.2-3.9 times higher than the straw-induced SOC sequestration rate, resulting in higher net GHG emissions[129] Accordingly, organic matter addition to rice paddies generally does not result in a net climate benefit[130]. Although crop yields often increase with increasing SOC content, yield increases level off at high SOC content[131], suggesting that in paddies with high SOC content, CH_4 emissions can be reduced through straw removal and by lowering manure addition while maintaining rice yield.

Paddy CH_4 emissions can also be reduced through optimizing the form, timing and application rate of organic matter, and through tillage practices (**Fig. 5**). The incremental effect of organic matter addition on CH_4 emissions typically increases with application rate[13], and is lower for composted manure than for fresh manure[10,132]. Adding green manure with higher C/N ratios generally leads to higher CH_4 emissions from rice fields[33,41]. However, the impact could be reduced through early application, for example during a preceding upland crop or in the fallow season rather than adding organic manure just before rice transplanting. This effect can be explained by aerobic decomposition before transplanting, which reduces substrate availability for methanogens once the field is flooded[49,133,134]. Returning straw during the non-rice growing season could potentially reduce global CH_4 emissions by 5.4 Tg yr^{-1} (ref [24]). Similarly, a short-term experiment in a rice-fallow system found that the climate benefit of soil C storage with straw retention can outweigh the increase in CH_4 emissions[10]. However, in many rice cropping systems (double rice and rice-wheat), there will be little time between flooded periods for straw to decompose under aerobic conditions.

Biochar-- charcoal produced through pyrolysis of biomass under O_2 deficient conditions—has further been proposed as an amendment to reduce GHG emissions from rice paddies. Biochar application reduces CH_4 emissions from rice paddies by 13% on average[40], possibly owing to reduced soil bulk density and increased soil pH, thereby increasing methanotrophic abundance and CH_4 oxidation rates[135–137]. The effect of biochar on methanogens is still unclear, although decreases in the abundance of methanogenic archaea have been reported[138].

Biochar application similarly reduces N_2O emissions by 22% in experiments $\geq$ 2 years[40], likely because biochar application generally raises soil pH, thereby reducing N_2O production ratios (N_2O /[N_2 + N_2O]) from nitrification. Biochar application might also reduce N_2O production from denitrification by stimulating the production and activity of the N_2O reductase enzyme[139,140]. Straw returned to the field in the form of biochar might reduce global CH_4 emissions from rice paddies by ~4.6 Tg yr^{-1} (ref [24]). However, biochar is currently applied on a limited area due to high cost[141] and the lack of functional and non-polluting biochar production equipment suitable for rice straw, the most abundant residue of rice production[141].

Organic matter management practices can thus strongly affect crop yield and CH_4 emissions, but these effects vary with SOC content. In soils with low SOC contents, compost manure addition, straw incorporation, and planting low C/N green manure are recommended to maintain rice yield and maximize reductions in GHG emissions. In soils with high SOC content, straw removal or straw return during the the upland crop seasons or fallow season (if unflooded) are preferred mitigation strategies.

[H2] Mineral N management

Mineral N fertilizers—which have increased in use[142]—also affect CH_4 emissions. These effects emerge through several mechanisms: increasing shoot and root growth, thereby substrate availability for methanogens[143]; reducing CH_4 consumption owing to CH_4 monooxygenase binding and reacting with NH_4^+ instead of CH_4 (ref [144]); stimulating the growth and activity of methanotrophs[145]; and increasing soil N availability[146,147].

The response of GHG emissions to mineral N fertilization varies with fertilizer rate and type. The effects of N input on CH_4 emissions decreases with increasing N rate[40,41], while N_2O emissions from rice paddies increase exponentially with increasing N application rates[8,40,148]. Considering N surplus and spatially explicit emission factors, optimizing N application rates could reduce N_2O emissions from rice paddies by ~43% without compromising rice yields[14]. The effect of N application on GWP did not vary with N application rates[40] and yield-scaled GHG emissions are most easily achieved at optimal N rates at which maximum yield is achieved[148]. In addition, ammonium sulfate application can reduce CH_4 emissions because SO_4^{2+} strongly suppresses CH_4 production[149,150].

Several practices focused on improving N use efficiency can also reduce GHG emissions. Enhanced-efficiency N fertilizers (controlled-release fertilizers, urease and nitrification inhibitors) decrease CH_4 emissions[151–153], mainly by increasing CH_4 oxidation[154]. Enhanced-efficiency N fertilizers also reduced N_2O emissions by 20-60% (ref [41,155,156]) by lowering the availability of N substrate for nitrification and denitrification[157,158]. Compared with broadcasting, sub-surface N application generally reduces CH_4 emissions from rice paddies by 13%on average by creating conditions that stimulate CH_4 oxidation rates, such as high Eh, and high availability of NO_3^- to act as electron acceptors[41,159–162]; the magnitude of N_2O emission reduction, however, is inconsistent[159,163]. Enhanced-efficiency N fertilizers and sub-surface N application also reduce total GHG emissions and generally increase rice yield[40,41], further emphasizing the potential to lower yield-scaled GHG emissions through improving N use efficiency.

N management can therefore affect CH_4 and N_2O emissions, as well as rice yield. The relative importance of these effects varies with N rate and fertilizer type. Thus, to increase rice yield and maximize reductions in GHG emissions with mineral N management, N rate at which maximum yield is achieved, enhanced-efficiency N fertilizers, and sub-surface N application should be utilised.

[H2] Tillage and crop establishment effects

Tillage practices affect soil bulk density, soil structure, moisture, temperature, and residue distribution, all of which could influence GHG emissions[164,165]. Thus, the type and timing of soil tillage operations can be altered to reduce GHG emissions (**Fig. 5**). No-till, one of the key components of conservation agriculture, reduces CH_4 emissions from rice paddies by 23% on average[40], likely through reduced labile C availability, soil Eh, and methanogen abundance[166–168]; no overall effect is observed on N_2O emissions and rice yield[40]. Tillage during the fallow season can further stimulate residue decomposition under aerobic conditions, thereby reducing GHG emissions during the rice growing season. For example, shifting tillage from spring (before transplanting) to winter (after harvest) reduced net GHG emissions by 46%-82% in a double rice system[169].

Crop establishment effects are also important. In Asia, rice is commonly grown by transplanting seedlings, although high labor cost has caused a shift towards other establishment forms[170,171].Other strategies include wet direct seeding (where seeds are broadcasted directly into a well-soaked or shallow flooded field) or dry direct seeding (where rice seeds are planted in a field, similar to maize or wheat; through rainfall or irrigation the rice establishes in a largely aerobic soil environment for the first month, after which the field is flooded). Both conventional-till and no-till direct seeded systems reduce CH_4 emissions by 40% to 60% compared to conventional-till transplanted rice, while direct seeded rice either increased N_2O emissions or the emission were similar[172]. Thus, provided that direct seeding equipment and technology to sustain rice yield are available, direct seeding, particularly dry direct seeding, shows great potential as a strategy to reduce GHG emissions.

However, these mitigation effects can be compounded by diversity in water mater management practices in direct-seeded systems[170]. For instance, comparing wet and dry direct seeding, yields are comparable between them; however, overall GWP is lower in dry direct seeded rice despite it having higher N_2O emissions[173,174]. On sandy loam soil, direct-seeded rice is often grown without flooding of the field, so that the soil conditions largely reflect those of aerobic rice systems. In these cases, the reduction in GHG emissions is mainly caused by the non-flooded water regime and not by the seeding method per se. In case of heavy rainfalls and clayey soils, however, direct seeding typically encompasses flooded conditions. Comparisons between transplanted and direct-seeded systems should also consider GHG emissions in flooded nurseries in transplanting systems[175]. The contribution of GHG emissions in flooded nurseries to annual emissions are still unknown. However, since rice nursery beds generally occupy less than 10% of the total field[175], the net difference in GHG emissions between direct-seeded and transplanted systems will be determined mostly by the water management during the seeding stage in direct-seeded systems.

The practice of harvesting a second rice crop from tillers originating from the stubble of a previously harvested crop--ratooning—is expanding in the US, China and Africa[176], and has important influences on GHG emissions. In such a system in the USA, CH_4 emissions during the ratoon crop accounted for almost 75% of total growing season emissions[177], attributed to straw from the first crop decomposing under flooded, anaerobic conditions during the ratoon crop. Yet, in China, CH_4 emissions are much lower in the ratoon rice season than in the first rice season owing to lower temperatures, shorter rice-growth periods, and lower C inputs from root residues and rhizodeposition[178,179]. Compared to the double rice system, the ratoon rice system reduces the area-scaled and yield-scaled GHG emissions in China[179].

Finally, CH_4 emissions from rice cropping systems also depend on whether and how rice is being rotated with other crops (rice-rice, rice-wheat, and rice-rape)[180–182]. For example, CH_4 emissions during the rice growing season were 61% higher in a rice-rape system than in a rice-wheat system[181], likely because the total C amount of rapeseed residues was much higher than that of wheat residues. In addition, a long-term rotation for upland crops can change the form of soil Fe, delaying the development of reductive soil conditions in the next rice season[183], thereby possibly reducing CH_4 emissions.

Tillage and crop establishment management thus affect a range of soil conditions, with consequences for CH_4 and N_2O emissions. To optimize rice yields while minimizing GHG emissions, conventional tillage during the fallow season and no-tillage during the rice season are generally recommended. However, these management practices require no-tillage transplanting equipment and technology to sustain rice yield, which might not always be available. No-tillage practices can be combined with direct seeding, if direct seeding equipment is available. Where thermal energy exceeds the requirement for single rice but is not enough for double rice, planting ratoon rice is an option.

[H2] Liming

Soil acidification, a key limiting factor for crop production, promotes the solubility and toxicity of aluminum (Al), manganese (Mn), and iron (Fe), causing nutrient deficiencies[184–186].The global lowland rice paddy area with such acid soils (where pH <5.5) is estimated at 0.32 million km^2 (ref [187]). Lime (limestone and dolomite) application is a common practice to alleviate soil acidification and improve crop yield[188,189]. However, because liming alters soil physio-chemical and biological properties, it can also affect GHG emissions[190–192].

In general, liming is thought to reduce CH_4 emissions from acidic rice paddies by ~20% (ref [187,193]).Several mechanisms drive this reduction, including the stimulation of soil microbial activity and organic matter decomposition under fallow conditions, thereby reducing substrate availability for methanogens[194]; increased rice root growth and root O_2 loss, reducing methanogenic growth and stimulating methanotrophic growth[194]; and reduced N_2O emissions, mainly by increasing the activity of N_2O reductase enzymes and shifting the soil microbial community towards bacterial dominance[187,193].

The mining, transport and application of lime requires energy, thereby causing indirect CO_2

emissions from fossil fuel burning. In addition, the dissolution of lime in soil also produces CO_2 (ref [186]). However, the GWP of these additional CO_2 emissions (97-102 kg CO_2 ha^{-1} yr^{-1}) are much lower however than the reduction of CH_4 emissions (1172 kg CO_2e ha^{-1} yr^{-1}) from acidic paddy soils[187]. Moreover, the production of some alternative liming materials such as steel slag require little additional CO_2 emissions, as they are by-product of steel manufacturing[195,196]. Liming of acidic rice paddies generally increases rice yields[197] and slightly increases SOC stocks[187]. Thus, liming can increase rice yields while reducing GHG emissions from acidic paddies.

Liming represents a relatively new approach to reduce CH_4 emissions from rice paddies with an increase rice yields in acid soils. Because liming material is relatively expensive, whether farmers will adopt this practice will likely depend on financial incentives (government funding).

[H2] Emerging mitigation practices

Multiple new techniques have emerged to reduce GHG emissions from rice paddies that have not yet been tested in broader field experiments. For instance, irrigation of oxygen-nanobubble water reduces CH_4 emissions owing to the oxygenation of shallow soil[198,199], as also achieved with, addition of oxygen-releasing chemicals (magnesium peroxide and calcium peroxide)[200] and oxygen-releasing biofertilizers (azolla and blue-green algae)[201,202]. Application of Fe(III) fertilizer can further reduce seasonal CH_4 by suppressing methanogenesis and stimulating anaerobic oxidization by Fe(III)[203]. Moreover, foliar application of kinetin and indole acetic acid can reduce CH_4 emissions, presumably by lowering root biomass, while increasing rice yield[204]. Cellulose acetate coated ethephon could also lower CH_4 emissions from rice paddies through reducing the abundance of active methanogens[205].

In addition to these applications, soil microbial communities can also be modified through natural selection and genetic engineering to reduce GHG emissions. For example, a one-time inoculation of cable bacteria reduced CH_4 emissions by 93% via increased soil sulfate levels through electrogenic sulfide oxidation[206]. Application of efficient CH_4 utilizing and plant growth promoting bacteria can reduce CH_4 emissions by 7-12% (ref [207]). Arbuscular mycorrhizal fungi inoculation also reduce N_2O emissions from rice paddies through increasing plant N uptake[208], as also achieved by inoculation of *nosZ*+ and non-genetically modified organism *nosZ*++ strains of *B. japonicum* at a field scale[209]. Inoculation with plant growth-promoting bacterium can also reduce N_2O emissions from cropland through reduction of ammonia oxidizing bacteria and increases in the abundance of N_2O-reducing bacteria[210].

[H2] Choosing effective mitigation practices

The most effective approaches to reduce GHG emissions vary with the temporal emission patterns (**Fig. 2**). For an early emission peak (often found in rice growing regions with high organic C input and with effective NCF practices), it is important to reduce substrate availability to methanogens or increase CH_4 oxidation at the beginning of the growing season. This need can be achieved through a reduction in organic C input such as straw and manure, or by switching from fresh manure to compost manure instead of fresh manure, especially in paddies with high SOC. Effective mitigation approaches also include the deep placement of N fertilizer, enhanced-efficiency N fertilizers, rice cultivars with high O_2 release, and dry direct seeding.

For two emission peaks (rice growing regions with high organic C input but without effective NCF practices), applying NCF during the late stage of rice season to reduce the late peak of CH_4

emissions is the most effective mitigation approach. In addition, reduced organic C input and deep placement of N fertilizer and enhanced-efficiency N fertilizers can help to minimize the first peak of CH_4 emissions.

For a late emission peak (rice growing regions with low SOC and organic C input), applying NCF during the late stage of rice season and selecting rice cultivars with low rhizodeposition are the most effective approaches. These approaches are successful given that rhizodeposition forms a key substrate source for methanogens in the later stage of the growing season[91–93].

Finally, for the bell-shaped temporal pattern (rice growing regions with continuous flooding and high temperature), NCF practices are the most effective approaches. However, in wet season rice or in lowlands, it is difficult to apply the NCF practices and CH_4 mitigation potential of NCF will be smaller[211]. In this case, deep placement of N fertilizer, enhanced-efficiency N fertilizers and selecting rice cultivars with low rhizodeposition are the most effective approaches.

[H1] Summary and future perspectives

Rice paddies are important sources of the powerful greenhouse gases CH_4 and N_2O, particularly in China and India, CH_4 produced during anaerobic decomposition of organic matter and N_2O a by-product of nitrification and denitrification (**Box 1**). Based on *in-situ* observations, global CH_4 emissions, N_2O emissions and yield-scaled GHG emissions from rice paddies average 283 kg CH_4 ha^{-1}, 1.7 kg N_2O ha^{-1}, and 0.9 kg CO_2e kg^{-1}, respectively. Global average yield-scaled CH_4 emissions have been reduced by 38-55% since ~1980, largely owing to rice yield increase and expansion of paddy drainage practices. These emissions are projected to increase under climate change scenarios, with field experiments suggesting that warming will increase global CH_4 emissions by 15-23%. Optimizing organic matter, water, and nitrogen management are the most promising avenues to reduce GHG emissions from global paddies while maintaining rice yields. However, the efficiency of mitigation approaches will depend on the local CH_4 emission patterns of rice growing seasons.

Despite advancing understanding, several key issues should be addressed in future research. Regional and global CH_4 emissions are currently associated with large uncertainties. To reduce this uncertainty, *in situ* GHG emission measurements from understudied regions should be expanded, particularly in low-latitude regions with high levels of rice production, such as the Philippines, Indonesia and Southern India. Data on the distribution of key agricultural practices known to affect GHG emissions scale should also be collected on the national or provincial level. Several discovered mechanisms that affect CH_4 emissions, such as the effects of rice plants on CH_4 oxidation rate and the activity of methanogens and methanotrophs have not yet been incorporated into leading GHG models such as DNDC, CH4MOD, DAYCENT, and DLEM[212–215].

The effects of climate change on GHG emissions from rice paddies vary with agricultural practices and local environmental conditions[19,69,73]. However, the number of climate change field experiments is still low, particularly in low-latitude regions[19,73]. Furthermore, most previous experiments were short-term (< 5 years), even though many factors that affect GHG emissions (plant adaptation, soil physical and chemical properties) might operate on longer-term scales[68,76,216,217]. Moreover, to improve the resilience of rice cropping systems to climate change, a great deal of effort has been directed towards germplasm development and improvement of agronomic practices (**Supplementary text**). Yet, the interaction effects of climate change and adaptation strategies on GHG emissions are still unknown. Thus, to accurately estimate the effects of climate change, more long-term experiments under climate change conditions covering typical soil and climatic regions and adaptation strategies are urgently needed.

Optimization of any single practice has limited potential of GHG mitigation and rice yield improvement (**Supplementary Table 5**). To increase rice yield while reducing GHG emissions, integrated agronomic management (cultivar, water, organic matter, nitrogen, and tillage) should be optimized[218], but there is limited assessment of integrated agronomic management on GHG emissions[218,219]. Indeed, individual agricultural practices interact in their effects on GHG emissions (cultivar×straw, N application×water management, water×straw)[22,220–222], but many of these interactions are still unclear. Future research should quantify these interactions and consider GHG emissions both during the rice season and the fallow season to determine the efficacy of agronomic management practices in mitigating GHG emissions.

While agricultural practices can affect CH_4 and N_2O emissions, they also affect soil respiration and lead to soil C change. Unfortunately, side-by-side comparisons of CH_4 and N_2O fluxes vis-à-vis C storage has seldom been investigated in long-term field experiments with a sufficient time horizon to determine the net effect of climate change and agricultural practices on the overall GHG balance[29,128–130]. Furthermore, previous field experiments and meta-analyses generally focus on direct GHG emissions and neglect indirect GHG emissions (GHG release from the production of nitrogen fertilizer, biochar, and compost manure). However, these indirect emissions can contribute substantially to total GHG emissions from agricultural systems[223,224]. For example, GHG emissions from the composting process accounts for 35% of total GHG emissions induced by compost manure application[224]. Thus, long-term experiments that compare changes in SOC vs. GHG emissions from rice paddies and life cycle assessments are urgently needed.

Quantification of GHG emissions from rice paddies often rely on meta-analyses. Yet, meta-analyses vary substantially in their methodological approach to weight the importance of individual effect sizes and to account for non-independence of effect sizes, strongly affecting results[225]. Although quality standards for meta-analyses in agronomy have been proposed[226], the adherence to such standards is still limited[227]. Rigorous assessment is thus needed to ensure the quality and credibility of meta-analyses[228]. Moreover, apparent inconsistencies in past meta-analyses might be resolved through so-called second order meta-analyses[229]. Importantly, literature included in meta-analyses are not always representative of real-world agricultural practices, as evident by inclusion of straw management practices reducing eCO_2-related CH_4 emissions by an order of magnitude[73]. New meta-analytic techniques and upscaling approaches are available that account for variation in environmental factors and management practices, including the meta-forest approach[217].

Finally, soil microorganisms are a key factor controlling GHG emissions from ecosystems[95] and thus, understanding the links between community composition and function is key. While measurement of the abundances and turn-over rates of soil microorganisms in rice paddies is done, the effects of climate change and agricultural practices on the soil microbial community composition has received less attention[15,32]. Advances in molecular technologies in soil microbiology provide an opportunity to unravel the functional linkage between GHG emissions and soil microorganisms which can be applied in to develop new mitigation strategies.

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Author contributions

Y.J. designed the concept for this Review. H.Q. and X.Z. collected the data. Y.J., H.Q., X.Z. Y.D., and K.J.v.G. wrote the first draft. All authors contributed to the manuscript content.

Competing interests

The authors declare no competing interests.

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Data availability

Data used are available in Supplementary Data 1 and 2.

Fig. 1 | *In situ* greenhouse gas flux measurements in rice paddies. **a**, Published estimates (**Supplementary Data 1**) of CH_4 emissions (n=269). The rice planting area is indicated in light grey. **b**, as in **a,** but for N_2O emissions (n=200). **c**, as in **a,** but for area-scaled GHG emissions from rice paddies (n=198). **d**, as in **a,** but for yield-scaled GHG emissions (n=185). In-situ observations indicate a high level of heterogeneity in annual CH_4, N_2O, area-scaled and yield-scaled greenhouse gas emissions.

Fig. 2 | CH_4 emission patterns throughout the rice growing season. **a**, hypothetical temporal pattern of relative CH_4 emissions (blue line) with an early emission peak, as observed in 232 in-situ sites (Supplementary Table 3), and mitigation priorities for this pattern. Blue shaded areas indicate when the soils were flooded **b**, as in **a,** but with two emission peaks (n=156). **c**, as in **a,** but with a late emission peak (n=137). **d**, as in **a,** but with bell-shaped pattern (n=48). Temporal emission patterns vary with climate and management practices throughout the growing season and can be broadly classified into 4 different types.

Fig. 3 | Effects of elevated CO_2 and warming on CH_4 and N_2O emissions. **a**, Published estimates of the effects of elevated eCO_2 (blue[74]) warming (green[19]) and their interaction (grey; **Supplementary Table 4**) on CH_4 emissions. The horizontal line indicates the median, the boundaries of the box the lower quartile and the upper quartile, and error bars the maximum and minimum values excluding outliers. **b**, As in **a,** but N_2O emissions (warming effects from ref [20]). Both elevated CO_2 and warming generally increase GHG emissions from rice paddies.

Fig. 4 | CH_4 emissions as affected by breeding strategies to increase rice yield. **a**, The effect of increasing harvest index on CH_4 emissions. **b**, the effect of increasing rice plant biomass on CH_4 emissions. + and – indicate positive and negative effects, respectively. Increasing harvest index and increasing rice plant biomass reduce CH_4 emissions in continuously flooded rice paddies and in paddies with high soil organic C contents, respectively.

Fig. 5 | Potential mitigation strategies. Overview of management practices in rice agriculture to achieve high yields and low GHG emissions**.**

Box 1: The key processes driving CH_4 and N_2O emissions from rice fields

CH_4 emissions from soils are the result of CH_4 production, oxidation, and transport processes.

[b1] CH_4 production: CH_4 is one of the end products of organic matter mineralization under anaerobic conditions (where Eh < −150 mV). Methane is produced by methanogens, which mainly belong to the domain Archaea, and include acetotrophic methanogens and hydrogenotrophic methanogens[95]. The sources of methanogenic substrates include soil organic carbon, root exudates, residues from the preceding crops and external organic matter addition[230,231].

[b1] CH_4 oxidation: CH_4 produced in rice paddies can be consumed by aerobic methanotrophs in the topsoil and rhizosphere where O_2 and CH_4 gradients overlap[95]. While oxidation rates are highly variable, microbial taxa have distinct preferences in terms of high O_2 and tolerating low CH_4 concentrations vs. favoring high CH_4 and tolerating low O_2 concentrations. In addition, anaerobic oxidation generally consumes 10–20% of CH_4 that is being produced by tapping alternative electron acceptors[232,233].

[b1] CH_4 transport: In rice paddies, CH_4 is transferred to the atmosphere via three pathways: ebullition (bubble formation), liquid phase diffusion, and transport through the aerenchyma of rice plants[90]. The CH_4 emitted through the rice plant during flooding can reach up to 90% of the total emissions[90].

Net N_2O emissions mainly result from microbial nitrogen transformations, that is, nitrification and denitrification[115,234].

[b1] N_2O production: N_2O is formed during NH_3 oxidation (nitrification, under aerobic conditions) as an intermediate product between NH_4^+ and NO_2^- or NH_2OH. N_2O is also an intermediate of denitrification (anaerobic conditions) -- the reduction of NO_3 to N_2. In flooded rice paddies, nitrification rates are often limited by O_2 availability, resulting in low N_2O emissions. High N_2O production is observed under alternate wetting and drying condition (50–80% water-filled pore space)[115].

[b1] N_2O consumption: N_2O can be reduced to N_2 by N_2O reductase (denitrification). The activity of N_2O reductase is sensitive to soil pH and O_2 concentrations. Low soil pH decreases the activity of N_2O reductase and then results in high $N_2O/(N_2O+N_2)$[235]. Above 80% water-filled pore space, N_2 rather than N_2O becomes the main product of denitrification[115].

[b1] N_2O transport: When the soil is flooded, N_2O emission occurs predominantly through the rice plants, while in the absence of floodwater, N_2O is released mainly via diffusion to the soil surface[236].

ToC Blurb

Rice paddies account for a large proportion of total agricultural methane and nitrous oxide emissions. This Review outlines the characteristics, changes and mitigation options for these emissions, highlighting the benefits of water and organic matter management.