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RESEARCH ARTICLE

Exploring strategies for agricultural sustainability in super hybrid rice using the food–carbon–nitrogen–water–energy–profit nexus framework

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Highlights

- A comprehensive food–carbon–nitrogen–water–energy–profit (FCNWEP) nexus framework was established to quantitatively assess the multi-dimensional sustainability of super hybrid rice production.
- Integrated crop management 2 (ICM2) with optimized nitrogen, dense planting, and alternate wetting–drying irrigation increased yield by 13–30% and profit by up to 115%.
- The integrated management system reduced carbon, nitrogen, water, and energy footprints per yield unit, while enhancing nitrogen and energy use efficiencies.

Abstract

The breakthrough in super hybrid rice yield has significantly contributed to China's and global food security. However, the inherent conflict between high productivity and environmentally sustainable agriculture poses substantial challenges. Issues such as water scarcity, energy crises, escalating greenhouse gas emissions, and diminishing farm profitability threaten long-term agricultural sustainability. In response, we applied a holistic food–carbon–nitrogen–water–energy–profit (FCNWEP) nexus framework to comprehensively assess the sustainability of distinct crop management strategies across three sub-sites in Central China. Field experiments were conducted in Hubei and Hunan provinces from 2017 to 2021 using a widely adopted elite super hybrid rice cultivar (Y-liangyou 900). Four crop management treatments were implemented: a control (CK, 0 kg N ha^{-1}), conventional crop management (CCM, 210–250 kg N ha^{-1}, 7:3 basal:mid-tiller fertilizer ratio), and two integrated crop management (ICM) treatments (ICM1, 180–210 kg N ha^{-1}, 5:2:3 basal:mid-tiller:panicle initiation fertilizer ratio; ICM2, 240–270 kg N ha^{-1}, 5:2:2:1 basal:mid-tiller:panicle initiation:flowering fertilizer ratio). Variables assessed included grain yield, carbon footprint, nitrogen footprint, water footprint, energy footprint, nitrogen use efficiency, and economic benefits. Our results showed significant yield variations, with ICM2 consistently outperforming CCM and ICM1 across all three sites. In Jingzhou, Suizhou, and Changsha, ICM2's grain yield was 30.2, 24.7, and 13.3% higher than CCM, respectively. Net profits under ICM2 exceeded those of CCM and ICM1 by 31.8 and 115.2% in Jingzhou, 32.2 and 109.9% in Suizhou, and 15.4 and 34.0% in Changsha, respectively. Integrated crop management, particularly ICM2, demonstrated improved nitrogen and energy use efficiency, leading to reduced carbon, nitrogen, water, and energy footprints. Overall, composite sustainability scores derived from the FCNWEP framework indicated that both ICM2 and ICM1 exhibited higher sustainability levels compared to CCM. This study provides valuable insights into practical management methodologies and offers recommendations for enhancing agricultural sustainability.

Keywords: super hybrid rice, ecological footprint, rice production, integrated crop management, sustainability

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1. Introduction

Rice (*Oryza sativa* L.) is one of the most critical staple crops globally, with over half of the world's population relying on it as a primary dietary source (FAO 2020). As the leading producer and consumer of rice, China cultivates 19% of global rice area and produces 28% of total output, which sustains over 65% of its population (FAO 2022). Super hybrid rice varieties have significantly increased yields in China, raising the national average from 6.2 to 7.4 t ha^{-1}, thereby contributing to enhanced food security (Deng *et al.* 2019).

Super hybrid rice is recognized as a pivotal factor in ensuring China's food security and strengthening grain self-sufficiency (Ma and Yuan 2015; Deng *et al.* 2019). However, this achievement comes with environmental trade-offs. The widespread cultivation and associated production processes generate considerable greenhouse gas emissions (Jiang *et al.* 2013) and entail high consumption of water, nitrogen fertilizers, and energy. Conventional approaches aimed at maximizing yield often result in adverse environmental impacts and resource depletion. Compounding these issues are climate change, declining water availability, energy-intensive farming practices, and economic pressures on farmers, all of which challenge sustainable development (Chaudhary *et al.* 2018; Gustafson *et al.* 2021; Liu *et al.* 2021, 2023). In this context, integrated crop management emerges as a promising strategy to reconcile productivity with sustainability. Implementing integrated practices — such as variety selection (Jiang *et al.* 2013), organic inputs, water management (Yan *et al.* 2009; Xia *et al.* 2016), tillage methods (Mooney and Sjogersten 2022), and nitrogen management (Zhang *et al.* 2013) — has demonstrated potential for reducing greenhouse gas emissions. Studies indicate that adopting novel rice cultivars, discontinuous irrigation, and residue removal can reduce average emissions by 24, 44, and 46%, respectively (Qian *et al.* 2023). Therefore, integrating high-yielding super rice varieties with sustainable, environmentally sound crop management practices is essential to minimize agriculture's ecological footprint and promote long-term sustainability.

In recent years, ecological footprint accounting has gained prominence as a tool for assessing the sustainability of human activities. The ecological footprint encompasses multiple dimensions, including carbon footprint (CF), nitrogen footprint (NF), water footprint (WF), and energy footprint (EF). These indicators are increasingly used by ecologists to estimate greenhouse gas emissions, water use, and energy consumption within agricultural systems. Recent research shows that highly efficient, low-carbon irrigation techniques can reduce energy use by up to 50% and CO_2 emissions by as much as 90% (Qin *et al.* 2024). A meta-analysis further reveals that plastic mulching can significantly reduce nitrogen loss in China, depending on crop type, nitrogen application rate, and local environmental conditions (Wang *et al.* 2020). Estimation of CF, NF, WF, and EF typically relies on the life cycle assessment (LCA) framework (Sharma *et al.* 2011; Ramaswami *et al.* 2017; Sadhukhan 2022), which evaluates natural resource consumption and environmental impacts across sectors, including agriculture, industry, and energy. A comprehensive understanding of these footprints is crucial for advancing sustainable and environmentally responsible agricultural practices.

Modern food production depends on anthropogenic inputs and the integration of various agricultural resources within a given land area. The conversion of solar energy into food energy involves complex interdependencies among food, carbon emissions, water, and energy (Vora *et al.* 2017). Increasing crop yields requires greater investments in water and energy (Smajgl and Ward 2013), which in turn drive energy demand for water extraction, distribution, and processing — all contributing to carbon emissions (Liu *et al.* 2015). Similarly, excessive nitrogen fertilizer use not only emits greenhouse gases (Choudhury and Kennedy 2005; Vora *et al.* 2017) but also leads to eutrophication and high energy input (Choudhury and Kennedy 2005). Thus, a comprehensive analysis of the relationships among food, carbon, nitrogen, water, energy, and economic returns is essential for sustaining and balancing food production systems.

Prior research has examined various aspects of rice cultivation, including water supply and demand patterns (Zhang *et al.* 2023), morphological and physiological traits (Deng *et al.* 2022a, b), and irrigation technologies (Mubeen *et al.* 2022). These studies have provided valuable insights for water resource management (Wang *et al.* 2018), the development of drought-resistant varieties (Luo *et al.* 2019), and the adoption of intermittent irrigation (Kang *et al.* 2022). Despite these contributions, a comprehensive evaluation within the food–carbon–nitrogen–water–energy–profit (FCNWEP) framework — incorporating carbon, nitrogen, water, and energy footprints along with economic sustainability — remains underexplored.

While existing studies have illuminated components of sustainable rice farming, there remains a notable gap in holistic assessments across the entire FCNWEP nexus. Integration across economic viability, resource efficiency, and yield performance is an area requiring further investigation. Addressing this gap is critical for gaining deeper insight into the challenges and opportunities associated with sustainable rice cultivation.

Therefore, we employed CF, NF, WF, and EF to assess whether integrated crop management — encompassing water, plant density, and nitrogen management — not only reduces environmental footprints in super hybrid rice but also maintains high grain yield. This study aims to address two key questions: (1) What changes occur in food production, economic returns, and environmental impacts — measured by carbon, nitrogen, water, and energy footprints — under different integrated management regimes? (2) How effectively do comprehensive FCNWEP-based indices support the sustainability claims of these integrated crop management strategies? Our objective is to critically evaluate management practices relative to local traditional methods and provide preliminary guidance to help farmers manage agricultural resources more sustainably.

2. Methods and materials

2.1. Experimental site and test materials

Field trials were conducted in Jingzhou (112°31′E, 30°21′N) from 2017 to 2021, in Suizhou (113°35′E, 31°58′N) from 2017 to 2019, Hubei Province, China, and in Changsha (113°28′E, 28°39′N) from 2017 to 2018, Hunan Province, China. The location of the study area and the three experimental sites is shown in Fig. 1. The sites experience hot summers and cool to cold winters, with variation in growing season precipitation and solar radiation. Mean temperatures during the rice-growing season were similar across locations, ranging from 21.1°C (minimum) to 32.1°C (maximum). Average annual growing season solar radiation was 2,023.7 MJ m^{-2} in Jingzhou, 2,149.2 MJ m^{-2} in Suizhou, and 1,930.5 MJ m^{-2} in Changsha. Average annual total precipitation was 622.5 mm in Jingzhou, 444.4 mm in Suizhou, and 383.8 mm in Changsha, respectively.

Surface soil samples were collected from the top 20 cm layer, and soil properties were averaged over five years in Jingzhou, three years in Suizhou, and two years in Changsha. The physicochemical characteristics of the soils at each site are presented in Table 1. The super hybrid rice cultivar Y-liangyou 900 (YLY900) was used across all sites. YLY900, an elite super hybrid variety, was developed from Y58S (female parent) and R900 (male parent) (http://www.ricedata.cn/variety/).

2.2. Experimental design

To facilitate management and mechanical transplantation, large-scale experiments were conducted with each treatment covering approximately 667 m^2. Across 10 experimental runs, four crop management approaches (Fig. 2) were established: control (CK), conventional crop management (CCM), integrated crop management 1 (ICM1), and ICM2. Each approach incorporated an integrated strategy involving fertilizer application, planting density, and water management. Experiments were conducted continuously over multiple years at the three sites using these four management strategies.

Pre-germinated seeds were sown in designated nursery beds, and seedlings were transplanted into field plots at two per hill, 28 to 32 days after sowing. Transplantation dates in Jingzhou were June 3, 2017; June 1, 2018; June 10, 2019; June 10, 2020; and June 1, 2021. In Suizhou, transplantations occurred on May 20, 2017; May 28, 2018; and May 29, 2019. In Changsha, transplanting took place on June 6, 2017, and June 7, 2018.

Urea was used as the nitrogen (N) source, calcium superphosphate for phosphorus (P), and potassium chloride for potassium (K). The N:P_2O_5:K_2O mass ratio was maintained at 2:1:2 across years and sites. The four crop management practices at the three sites were as follows: CK received no fertilizer. Planting densities were 30 cm×18 cm in Jingzhou, 20 cm×20 cm in Suizhou, and 24 cm×20 cm in Changsha. In CCM, total N rates were 210 kg N ha^{-1} in Jingzhou and Changsha and 250 kg N ha^{-1} in Suizhou.

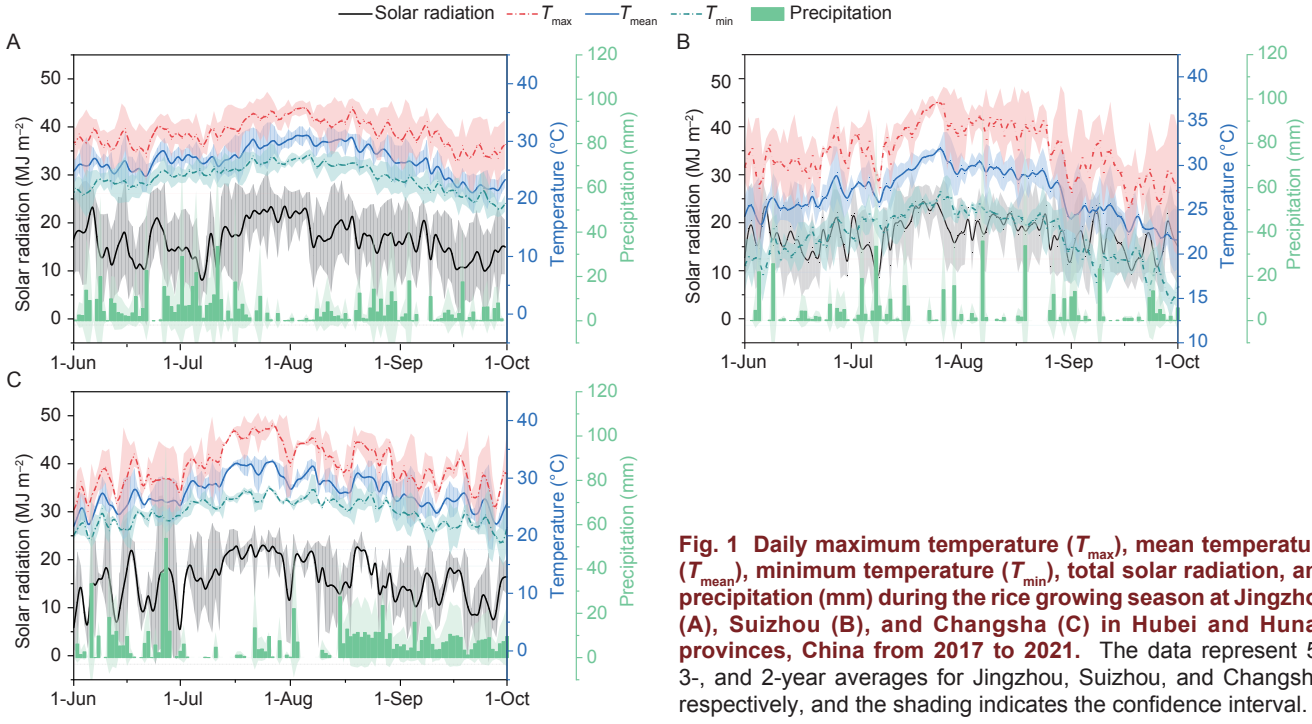


Fig. 1 Daily maximum temperature (T_{max}), mean temperature (T_{mean}), minimum temperature (T_{min}), total solar radiation, and precipitation (mm) during the rice growing season at Jingzhou (A), Suizhou (B), and Changsha (C) in Hubei and Hunan provinces, China from 2017 to 2021. The data represent 5-, 3-, and 2-year averages for Jingzhou, Suizhou, and Changsha, respectively, and the shading indicates the confidence interval.

Table 1 Soil physicochemical properties in the 0–20 cm soil layer across different test sites in Hubei and Hunan provinces, China

Site	Soil pH	Soil organic matter (g kg^{-1})	Total N (g kg^{-1})	Available P (mg kg^{-1})	Available K (mg kg^{-1})
Jingzhou	7.34	15.76	3.56	22.87	82.40
Suizhou	6.52	20.30	3.82	27.20	145.50
Changsha	5.93	15.40	3.25	20.30	116.25

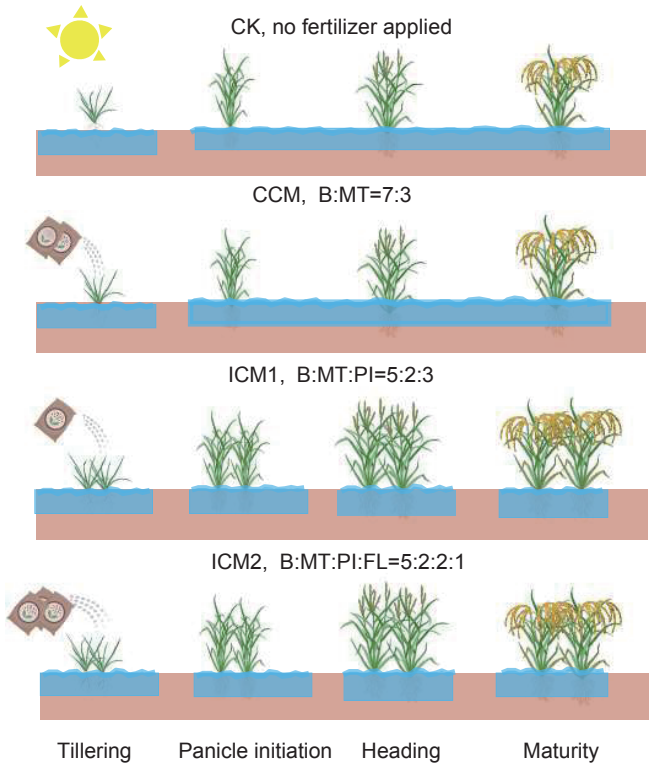


Fig. 2 Four crop management practices in rice production across three sites from 2017 to 2021. CK, the control; CCM, conventional crop management in the local area; ICM1 and ICM2, integrated crop managements 1 and 2, respectively; B, basal fertilizer application; MT, mid-tillering stage; PI, panicle initiation stage; FL, flowering stage.

Seventy percent of N was applied as basal fertilizer (3 days before transplanting), and 30% at mid-tillering (7 days after transplanting). Phosphorus fertilizer was applied entirely as basal in all treatments and sites. Potassium fertilizer was applied once as basal in CCM. The transplanting density in CCM matched that of CK.

In ICM1, total N rates were 195, 210, and 180 kg N ha^{-1} in Jingzhou, Suizhou, and Changsha, respectively. Nitrogen was applied at 50, 20, and 30% during basal (B), mid-tillering (MT), and panicle initiation (PI) stages, respectively. Phosphorus was split equally between B and PI stages (1:1 ratio). Planting densities were 30 cm×16 cm in Jingzhou, 16.7 cm×20 cm in Suizhou, and 24 cm×16 cm in Changsha. In ICM2, 270 kg N ha^{-1} was applied in Jingzhou and Suizhou, and 240 kg N ha^{-1} in Changsha, with a 5:2:2:1 split at B, MT, PI, and flowering (FL) stages. The potassium application schedule and transplanting density in ICM2 were identical to those in ICM1.

After transplanting, all four treatments were maintained under shallow flooding until 5 days before panicle initiation. Following a 5-day drainage period, all treatments were re-flooded at panicle initiation. CK and CCM remained continuously flooded until 10 days before harvest, then drained. ICM1 and ICM2 employed alternate wetting and drying (AWD) irrigation (Qin *et al.* 2013) until 10 days before harvest, followed by final drainage. To prevent visible cracking, fields under AWD were irrigated to a depth of 3.0 cm, allowed to dry, and re-irrigated to the same depth. Water meters were used to accurately record irrigation volumes at all three locations. Throughout the growth cycle, pests and diseases were managed chemically. Cultivation methods were consistent across locations and years.

2.3. Measurement of nitrogen use efficiency and grain yield

Twelve plants were sampled based on average tiller number per plot, with three replicates per plot. Samples were transported to the laboratory at maturity. Roots were removed, and plants were separated into stems, leaves, and spikes, placed in paper bags, and subjected to a 105°C heat treatment for 15 min, followed by drying at 80°C to constant weight. Grain yield (GY) was measured in a 5 m^2 area per plot and adjusted to 0.14 g H_2O g^{-1}.

Additionally, 12 diagonal hills within the measurement area were sampled to assess panicle number, spikelets per panicle, grain filling, and 1,000-grain weight, as detailed in Deng *et al.* (2022a). Partial factor productivity of applied nitrogen (PFP_N) was calculated as grain yield divided by total N fertilizer input. Nitrogen agronomic efficiency (AE_N) was determined as the increase in grain yield due to nitrogen application divided by the amount of nitrogen applied (Fageria and Baligar 2005).

2.4. Evaluation indicators of rice production

The carbon footprint, nitrogen footprint, water footprint, energy footprint, and economic value were calculated for each scenario (Fig. 3). Given the spatial heterogeneity of the sites, regional averages were computed using the inverse distance weighted averaging method.

Calculation of carbon footprint of rice production The greenhouse gas (GHG) emissions analyzed in this study include: (1) manufacturing, storage, transport, and application of agricultural inputs such as seeds, fertilizers, and pesticides; (2) fuel oil consumption during machinery operations (e.g., tillage, irrigation, harvesting), excluding emissions from equipment manufacturing; and (3) GHG emissions from paddy fields during the rice-growing season. Emissions from human respiration, rice respiration, and residue decomposition were excluded, as the latter two involve carbon sequestration during plant growth.

The global warming potential (GWP, kg CO_2-eq ha^{-1}) was used to quantify the climate impact of these emissions (Chen *et al.* 2020) and was calculated as follows:

$$GWP_{total}=GWP_{CO_2}+GWP_{N_2O}+GWP_{CH_4} \quad (1)$$

$$GWP_{CO_2}=\sum(AI_j\times\sigma_j) \quad (2)$$

$$GWP_{N_2O}=T_{N_2O}\times 298 \quad (3)$$

$$GWP_{CH_4}=T_{CH_4}\times 25 \quad (4)$$

where agriculture inputs (AI) denotes various AI, including fertilizer, seeds, pesticides, total mechanical power, electricity for irrigation, and diesel used in agricultural operations; j represents the jth production input in the rice production process; σ_j represents the GHG emission coefficient (kg CO_2-eq $unit^{-1}$) for the jth production material, following Chen *et al.* (2020) (Appendix A). T_{N_2O} represents the CO_2 equivalent (kg CO_2-eq ha^{-1}) derived from direct and indirect N_2O emissions

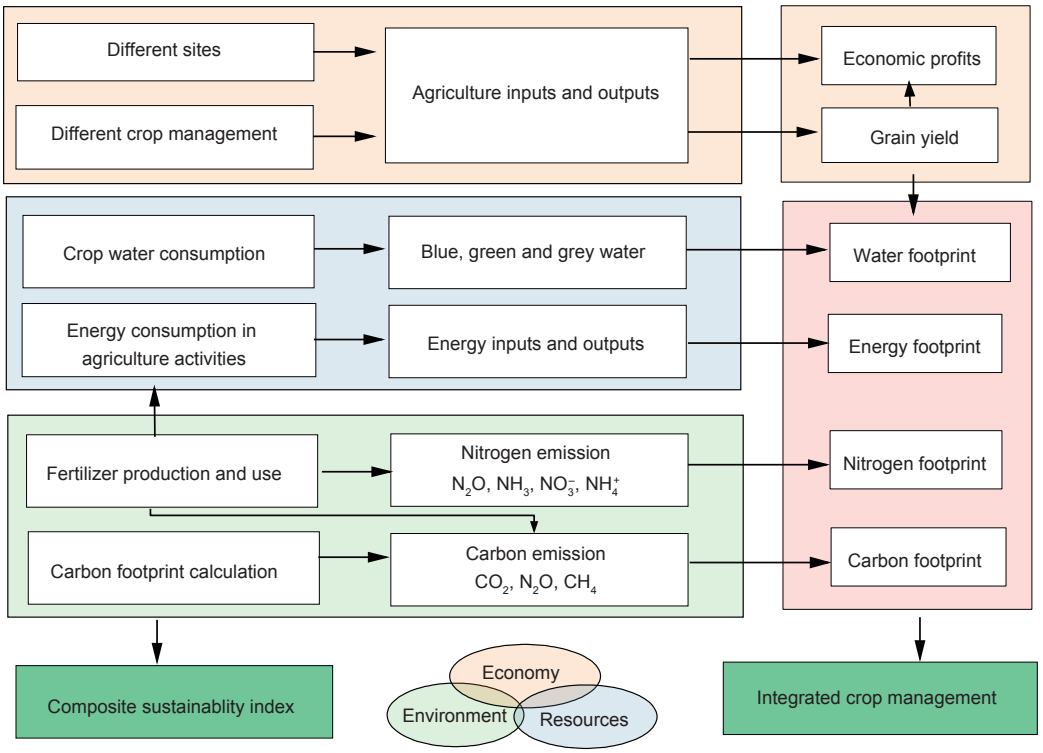


Fig. 3 Sustainability assessment framework integrating resource consumption, environmental impact, and food-related economic benefits.

resulting from nitrogen fertilizer application. Direct N_2O emissions from N fertilizer application were estimated using the relationship between N_2O emissions and N surplus (Zhou L Y *et al*. 2023; Zhou Y J *et al*. 2023). Indirect N_2O emissions in rice fields include those arising from atmospheric deposition of N volatilized as NH_3 and NOx, as well as emissions generated from N leaching and runoff. The calculation of indirect N_2O emissions in agricultural systems follows the 2006 IPCC Guidelines (IPCC 2006). T_{CH_4} represents the CO_2 equivalent (kg CO_2-eq ha^{-1} yr^{-1}) derived from CH_4 emissions during rice production, with the calculation method referenced from Chen *et al*. (2020). Using 100-year GWP estimates, the GWP conversion factors for CO_2, N_2O, and CH_4 are 1, 298, and 25, respectively (IPCC 2006; Chen *et al*. 2020). This study estimated the grain yield product CF in kg CO_2-eq t^{-1}, calculated as follows:

$$CF=GWP_{total}/\text{Grain yield} \quad (5)$$

Nitrogen footprint calculation of rice production In this study, the eutrophication potential was assessed to quantify the release of nitrogen reactive (Nr) compounds into the atmosphere, water, and soil. Furthermore, the environmental impacts of Nr emissions and losses associated with rice production processes were evaluated in accordance with the ISO 14044:2006 principles (ISO 14044 2006). The farm-level nitrogen footprint (FNF, kg N-eq ha^{-1}) and the grain yield-based nitrogen footprint (NF, kg N-eq t^{-1}) were estimated using eqs. (5)–(10). These metrics are calculated as follows:

$$FNF=NE_{inputs}+EV_{NH_3}+EV_{N_2O}+NL_{NO_3^-}+NL_{NH_4^+} \quad (6)$$

$$NF=FNF/\text{Grain yield} \quad (7)$$

where NE_{inputs} denotes the calculation of aggregated nitrogen losses associated with agricultural inputs and applications, computed by multiplying the actual usage amounts of various inputs by their respective emission factors. These losses are considered indirect and are detailed in Appendix A, based on Chen *et al.* (2020).

Calculation of energy footprint of rice production The energy input in rice production comprises eight components: irrigation water, machinery, diesel, electricity, labor, seeds, fertilizers, and pesticides (Jin *et al.* 2023a). Energy output includes rice grains and straw. The energy content of both input materials and output products is determined by multiplying their quantities by corresponding energy equivalents (Jin *et al.* 2023b). Total energy input is the sum of all individual input energies, while total energy output is the sum of all output energies. The energy equivalents for these materials and products were shown in Appendix B. Net energy, energy use efficiency, energy productivity, and energy profitability in rice production are calculated using both component-specific and system-wide energy inputs and outputs. These metrics are derived following equations from Pirdashti *et al.* (2015) and Singh *et al.* (2019). The energy footprint (EF) is computed according to the methodology outlined by Jiang *et al.* (2022).

$$\text{Net energy}=\text{Energy out}-\text{Energy input} \quad (8)$$

$$\text{Energy use efficiency}=\text{Energy out}/\text{Energy input} \quad (9)$$

$$EF=\text{Energy input}/\text{Yield} \quad (10)$$

$$\text{Economic productivit}=\text{Total yield}/\text{Costs} \quad (11)$$

Calculation of water footprint of rice production The water footprint of rice production quantifies the total freshwater consumed and required to dilute pollutants

generated during the crop's growth cycle (Chapagain and Hoekstra 2011; Jin *et al.* 2023a). It comprises three components: the blue water footprint (WF_{blue}), the green water footprint (WF_{green}), and the gray water footprint (WF_{gray}) (Luo *et al.* 2019). Blue water refers to the volume of surface and groundwater consumed, typically accounted for as irrigation. Green water denotes the portion of effective precipitation utilized by the crop without contributing to runoff. Gray water represents the volume of freshwater necessary to assimilate pollutants from agricultural activities — such as nutrient leaching or pesticide runoff — into natural water bodies while maintaining compliance with environmental water quality standards (Zhou L Y *et al.* 2023; Zhou Y J *et al.* 2023).

$$WF_{total}=WF_{blue}+WF_{green}+WF_{gray} \quad (12)$$

$$WF_{blue}=W_{blue}/\text{Grain yield} \quad (13)$$

$$WF_{green}=W_{green}/\text{Grain yield} \quad (14)$$

$$WF_{gray}=\frac{N\times\alpha}{C_{max}\times\text{Grain yield}} \quad (15)$$

where W_{blue} refers to the blue water demand in rice production, equivalent to the volume of irrigation water applied (m^3 ha^{-1}) during the rice growth period in this study. W_{green} denotes the green water demand in rice production, corresponding to the effective precipitation (m^3 ha^{-1}) during the same period. Given the limited rainfall during the rice growing season, effective precipitation was assumed to equal total rainfall in this analysis. In estimating the grey water footprint, only nitrogen fertilizer-induced pollution loading to water bodies was considered; contributions from other fertilizers and pesticides were excluded. To ensure methodological consistency, a natural background concentration of 0 for nitrogen in receiving waters was assumed, and the grey water footprint was calculated using the critical dilution approach (Ridoutt and Pfister 2013). The variable N represents the rate of nitrogen application (kg ha^{-1}). α denotes the proportion of nitrogen entering water bodies *via* leaching and surface runoff, approximated as 0.1 based on Chapagain and Hoekstra (2011). C_{max} indicates the maximum allowable nitrogen concentration in the receiving water body, set at 10 mg L^{-1} (He *et al.* 2024).

Calculation of economic benefits of rice production Costs encompass total agricultural input expenditures, with the unit cost of inputs (USD ha^{-1}) estimated by multiplying the quantities of various inputs — such as land leasing fees, fertilizers, pesticides, labor, machinery, and seeds — by their respective market prices (USD U^{-1}) (Zhou L Y *et al*. 2023). Income (USD ha^{-1}) is calculated by multiplying the total grain yield of the rice system by the prevailing market price. Price data were obtained from authoritative sources: the National Bureau of Statistics of China (https://www.stats.gov.cn/) and the National Food and Strategic Reserves Administration of China (http://www.lswz.gov.cn), ensuring dataset reliability. All cultivation and management practices were kept constant across years and sites in this experiment. Unit prices for inputs and outputs are used to assess the economic performance of rice production (Appendix C). The calculation methods for net profit (NP) and economic productivity are as follows:

$$NP=\text{Income}-\text{Costs} \quad (16)$$

$$\text{Gross margin}=NP/\text{Costs} \quad (17)$$

$$\text{Economic productivit}=\text{Total yield}/\text{Costs} \quad (18)$$

Rice production system sustainability assessment framework To evaluate the six domains encompassing food production (referred to as grain yield in this study), carbon footprint, nitrogen footprint, water footprint, energy footprint, and net profits, a composite sustainability index was calculated using the FCNWEP nexus framework, which has been established in recent research (Nhamo *et al.* 2020; Jiang *et al.* 2022; Jin *et al.* 2023b).

To enable comparison across indicators measured in different units, normalization was applied to transform values onto a uniform scale from 0 to 100. Given that lower ecological footprints are environmentally favorable, whereas higher production and profitability are economically beneficial, CF, NF, WF, and EF serve as key metrics for environmental impact assessment. Higher values of these indicators generally correspond to greater environmental burden. In contrast, increased crop yields and economic returns are advantageous for agricultural systems. Accordingly, two distinct min–max normalization methods were applied: one for ecological footprint indicators (eq. (19)) and the other for profitability indicators (eq. (20)). The sustainability score was derived as the mean of the six standardized indicators, with equal weights assigned to ensure equivalent importance across all domains. The evenness score (ES) was computed from the four normalized environmental indicators (Liu *et al.* 2020; Wang *et al.* 2022).

$$S_{CNWE}=\frac{S_{max}-S_i}{S_{max}-S_{min}} \quad (19)$$

$$S_{FP}=\frac{S_i-S_{min}}{S_{max}-S_{min}} \quad (20)$$

$$S_{FCNWEP}=(S_F+S_C+S_N+S_W+S_E+S_P)/6 \quad (21)$$

$$ES\ (\%)=\frac{A_i}{\pi\times(L_i/2\pi)^2}\times100 \quad (22)$$

$$L_i=2|(r_{max}-r_{min}|+\sum_{i=1}^{n}2\pi w_i r_i \quad (23)$$

$$A_i=\sum_{i=1}^{n}\pi w_i r_i^2 \quad (24)$$

where S_{CNWN} and S_{FP} represent the normalized values of carbon, nitrogen, water, energy, food, and net profit indicators, respectively. S_{max} and S_{min} denote the uppermost and lowermost limits of each indicator. S_{FCNWEP} represents the composite sustainability index, ranging from 0 to 100. *ES* is the evenness score, defined as the ratio of the aggregate area of six indicators in a radar chart to the area of a circle with an equivalent perimeter L_i, which corresponds to the most uniform distribution of the indicators. *ES* ranges from 0 to 100 and decreases as disparity among the six indicators increases. In this study, w_i denotes the weight assigned to each indicator and is set at 1/6. Equal weighting was applied to avoid subjective bias and to ensure comparable importance among environmental and economic indicators. r_i represents the value of each indicator used as the radius. The twofold magnitude of the difference between r_{max} and r_{min} corresponds to the portion of the boundary not enclosed by the total arc length calculated as $2\pi w_i r_i$.

2.5. Data analysis

Raw data from field experiments, including recordings, visualizations, and image outputs, were processed using

Microsoft Excel 2020, Origin 2024 Pro, and ArcGIS Pro 10.5. Analysis of variance was conducted using SPSS software (version 26.0), with least significant differences (LSD) assessed at a significance level of 0.05 unless otherwise specified.

3. Results

3.1. Grain yield

Significant differences in grain yield were observed among the four crop management practices, with ICM2 consistently achieving higher yields across all sites (Fig. 4; Table 2). In Jingzhou, the annual average yield under ICM2 reached 11.19 t ha^{-1}, representing increases of 30.2 and 15.0% compared to CCM and ICM1, respectively. Additionally, the annual average yield under ICM1 was 13.2% higher than that under CCM. In Suizhou, ICM2 achieved an annual average yield of 11.64 t ha^{-1}, exceeding CCM and ICM1 by 24.7 and 13.6%, respectively. Similarly, in Changsha, ICM2 attained an annual average yield of 10.87 t ha^{-1}, reflecting increases of 13.3 and 9.4% over CCM and ICM1, respectively.

3.2. Carbon footprint of rice production

Across all sites and under different crop management practices, ICM2 exhibited the highest GWP (Fig. 5; Table 2). In Jingzhou, the GWP of ICM2 reached 5841.06 kg CO_2-eq ha^{-1}, representing increases of 7.1 and 6.8% compared to ICM1 and CCM, respectively. The GWP of ICM1 was slightly lower than that of CCM. Significant differences in GWP were also observed in Suizhou and Changsha. In Suizhou, ICM2's GWP was 5,369.93 kg CO_2-eq ha^{-1}, showing increases of 6.5 and 3.3% relative to ICM1 and CCM, respectively. In Changsha, the GWP of ICM2 increased by 7.0 and 4.8% compared to ICM1 and CCM, respectively.

With respect to production CF, both ICM1 and ICM2 showed lower values than CK and CCM (Fig. 5). The annual average CF of ICM2 was 522.28, 462.04, and 538.69 kg CO_2-eq t^{-1} in Jingzhou, Suizhou, and Changsha, respectively. Specifically, compared to ICM1 and CCM, the CF of ICM2 was reduced by 7.1 and 18.5% in Jingzhou, and by 6.2 and 17.2% in Suizhou. In Changsha, both ICM1 and ICM2 exhibited significantly lower CF than CCM, although the difference between ICM1 and ICM2 was not significant.

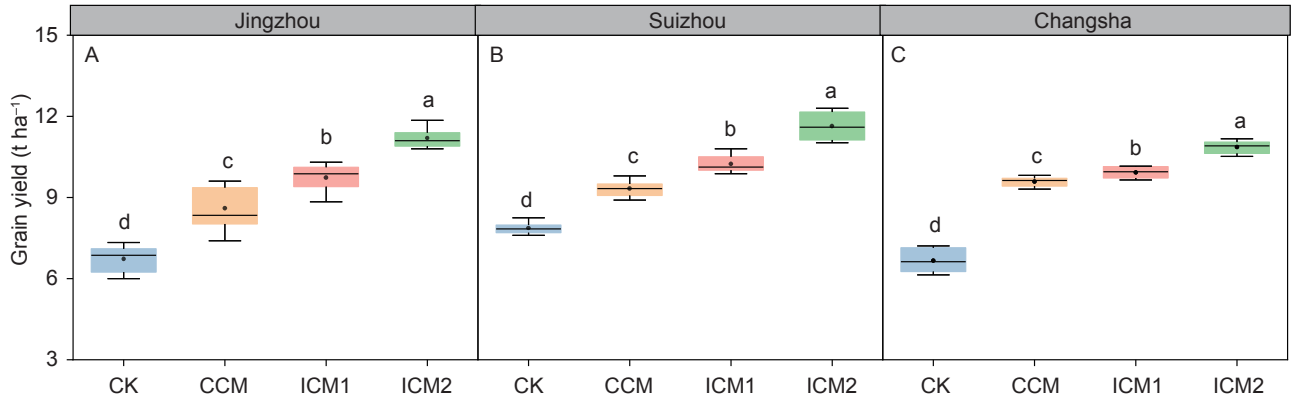


Fig. 4 Average grain yield under four crop management practices in Jingzhou (A), Suizhou (B), and Changsha (C) in Hubei and Hunan provinces, China from 2017 to 2021. Boxplots show the distribution of rice yields across locations; the solid black dot indicates the mean, and the horizontal line within each box represents the median. The box boundaries correspond to the 25th and 75th percentiles, while the whiskers extend to the 10th and 90th percentiles, respectively. CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. Bars indicate SD (n=15, 9, and 6 for Jingzhou, Suizhou, and Changsha, respectively). Means sharing the same letter within each site are not significantly different according to LSD test at P=0.05.

Table 2 Analysis of variance (ANOVA) for F-values of GY, nitrogen efficiency metrics (AE_N and PFP_N), ecological footprints (CF, NF, WF, and EF), and NP under four crop managements across sites in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021[1)]

Site	ANOVA	GY	AE_N	PFP_N	CF	NF	WF	EF	NP
Jingzhou	Year (Y)	100.43**	9.73**	595.76**	151.52**	64.58**	97.35**	329.03**	80.47**
	Treatment (T)	2,076.86**	304.93**	808.63**	515.48**	1,319.32**	894.97**	1,262.51**	72.50**
	Y×T	13.69**	11.93**	43.89**	17.01**	25.46**	16.24**	656.12**	10.72**
Suizhou	Y	88.67**	1.26 ns	0.76 ns	61.85**	83.53**	63.47**	3.68*	85.65**
	T	1,666.67**	219.06**	787.69**	357.65**	2,596.14**	726.48**	48.51**	641.28**
	Y×T	15.76**	5.75**	10.60**	10.80**	25.10**	10.48**	6.42**	14.65**
Changsha	Y	104.04**	0.05**	0.49 ns	96.26**	40.67**	115.15**	36.60**	106.58**
	T	1,483.31**	160.34**	413.91**	360.68**	1,516.96**	689.32**	317.42**	644.56**
	Y×T	7.46**	2.49 ns	0.61 ns	24.39**	4.83*	26.50**	1.83 ns	8.99**

[1)] GY, grain yield; AE_N, nitrogen agronomic efficiency; PFP_N, partial factor productivity of applied nitrogen; CF, carbon footprint of production; NF, nitrogen footprint of production; WF, water footprint; EF, energy footprint; NP, net profit.
** and *, significant at P<0.01 and P<0.05, respectively; ns, non-significant.

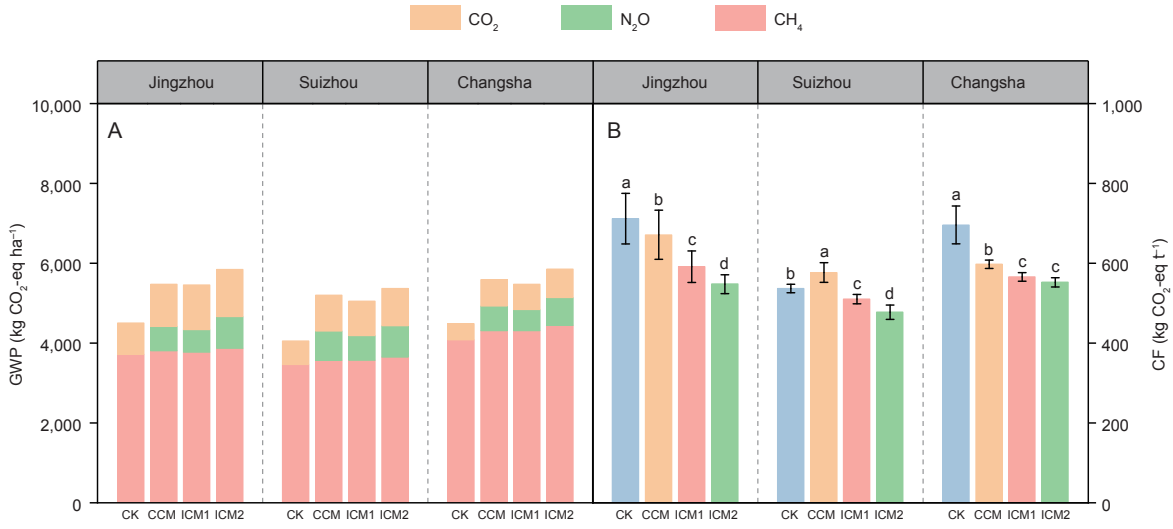


Fig. 5 Average global warming potential (GWP) (A) and carbon footprint of production (CF) per unit production (B) under four crop management systems in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China, from 2017 to 2021. CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. Bars indicate SD (n=15, 9, and 6 for Jingzhou, Suizhou, and Changsha, respectively). Means sharing the same letter within each column are not significantly different according to LSD test at P=0.05.

The CF of ICM2 and ICM1 decreased by 7.6 and 5.4%, respectively, compared to CCM.

Notably GWP_{CH_4}, contributed the largest proportion — over 60% — to the total GWP. However, variations in GWP among treatments were primarily driven by N_2O emissions, while the contributions of CH_4 emissions and agricultural inputs to GWP remained relatively stable. Compared to ICM1 and CCM, the GWP_{N_2O} under ICM2 increased by 29.2 and 20.4% in Jingzhou, 20.8 and 4.6% in Suizhou, and 24.7 and 9.1% in Changsha, respectively.

3.3. Nitrogen footprint and nitrogen use efficiency of rice production

Nr emissions under ICM2 were 274.10, 273.92, and 243.37 kg N-eq ha^{-1} in Jingzhou, Suizhou, and Changsha, respectively (Fig. 6-A and B). Relative to ICM1 and CCM, Nr emissions under ICM2 were 38.3 and 28.5% higher in Jingzhou, 28.5 and 8.0% higher in Suizhou, and 33.3 and 14.3% higher in Changsha, respectively.

In Jingzhou, the NF of ICM1 (20.41 kg N-eq t^{-1}) was 18.3% lower than that of CCM (Fig. 6-B). Although the NF of ICM2 was 20.1% higher than ICM1, it did not differ significantly from CCM. In Changsha, the NF of ICM1 (18.40 kg N-eq t^{-1}) was 17.2% lower than CCM, and while ICM2 had a 21.8% higher NF than ICM1, it was not significantly different from CCM. In Suizhou, both ICM2 and ICM1 showed notably lower NF than CCM (20.83 kg N-eq t^{-1}), by 13.4 and 23.5%, respectively. The composition of NF across sites and treatments primarily included EV_{NH_3}, $NL_{NO_3^-}$, and $NL_{NH_4^+}$ (Fig. 6-A). Specifically, compared to ICM1 and CCM, EV_{NH_3} under ICM2 increased by 35.5 and 28.6% in Jingzhou, 28.6 and 8.0% in Suizhou, and 33.3 and 14.3% in Changsha, respectively.

Significant differences in AE_N were observed among treatments, with consistent trends across years and locations (Fig. 6-C). Compared with CCM, AE_N under ICM1 and ICM2 increased by 72.8 and 85.6% in Jingzhou, 93.7 and 139.3% in Suizhou, and 30.3 and 25.8% in Changsha, respectively. On average, ICM1 and ICM2 enhanced AE_N by 63.7 and 77.6% relative to CCM. ICM1 consistently achieved the highest PFP_N across all sites, exceeding CCM and ICM2 by 22.4 and 19.9%, respectively (Fig. 6-D).

3.4. Water footprint of rice production

The water footprint of rice varied significantly across the four management practices at the three sites (Fig. 7; Table 2). CK consistently exhibited the highest water footprint, with values of 2,015.5, 1,729.9, and 2,033.8 m^3 t^{-1} in Jingzhou, Suizhou, and Changsha, respectively. Notably, ICM2 consistently demonstrated a reduced water footprint compared to both CCM and ICM1 across all sites.

In terms of water footprint composition, all four integrated management practices across regions showed WF_{blue} as the dominant component, accounting for more than 60% of the total. In contrast, WF_{gray} was the highest under ICM2, averaging 18.3%. Notably, ICM1 achieved a significant 8.7% reduction in water footprint relative to CCM.

3.5. Energy footprint of rice production

The energy footprint under the four crop management practices at the three sites were shown in Fig. 8. The energy footprint under CCM was the highest. Compared to CCM, the energy footprint under ICM1 and ICM2 decreased by 13.5 and 11.2% in Jingzhou, 17.6 and 15.6% in Suizhou, and 11.7 and 3.5% in Changsha, respectively.

The energy flows and efficiency of the four rice management practices across the three sites were shown in Fig. 9. Excluding CK, increased energy output led to higher net energy and improved energy use efficiency under different crop managements. Lower energy input in CK also enhanced its energy use efficiency and productivity.

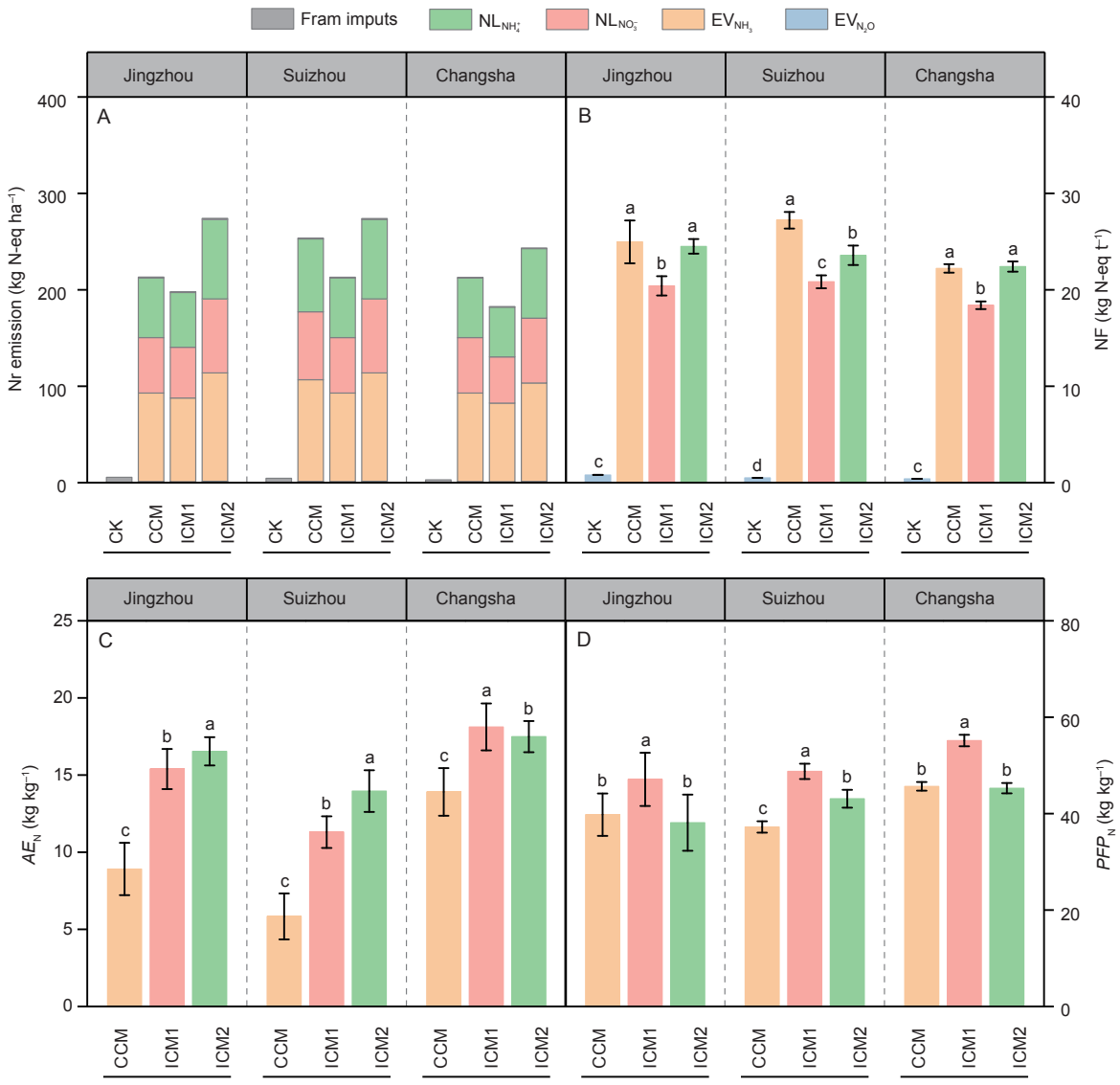


Fig. 6 Average values of Nr emission (A), production nitrogen footprint (NF) (B), nitrogen agronomic efficiency (AE_N) (C), and partial factor productivity of applied N (PFP_N) (D) under four crop management systems in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021. CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. Bars indicate SD (n=5, 9, and 6 in Jingzhou, Suizhou, and Changsha, respectively). Means sharing the same letter within each column do not differ significantly based on the LSD test at P=0.05.

Significant differences in net energy were observed among the four practices. Under ICM2, net energy was 9.2 and 25.5% higher than under ICM1 and CCM in Jingzhou, respectively. Similar increases were observed in Suizhou (9.6 and 24.4%) and Changsha (12.9 and 20.8%). For energy use efficiency, CK ranked highest, followed by ICM2, ICM1, and CCM. The energy output of ICM2 and ICM1 was 6.8 and 14.6% higher than CCM in Jingzhou, respectively, with similar trends in Suizhou (15.9 and 22.2%) and Changsha (9.2 and 14.9%). Energy productivity of ICM1 was 7.6 and 14.5% higher than ICM2 and CCM in Jingzhou, respectively, with comparable patterns in Suizhou (5.7 and 22.3%) and Changsha (4.8 and 15.1%).

3.6. Economic benefits of rice production

The annual average income under ICM2 was 4,607, 4,789, and 4,471 USD ha^{-1} in Jingzhou, Suizhou, and Changsha, respectively (Table 3). The corresponding annual average cost was 3,014, 3,270, and 2,936 USD ha^{-1}. Compared to ICM1 and CCM, net profit under ICM2 increased by 31.8 and 115.2% in Jingzhou, 32.2 and 109.9% in Suizhou, and 15.4 and 34.0% in Changsha, respectively. The rice gross margin under ICM2 was the highest across the three sites. Relative to CCM and ICM1, the gross margin of ICM2 increased by 47.7 and 31.9%, respectively. For economic productivity, ICM2 and ICM1 showed increases of 103.9 and 63.4% in Jingzhou, 100 and 60.9% in Suizhou, and 26.8 and 17.1% in Changsha, respectively, compared to CCM.

3.7. FCNWEP composite sustainability and evenness

The FCNWEP index revealed consistent patterns in sustainability scores across the three sites (Fig. 10).

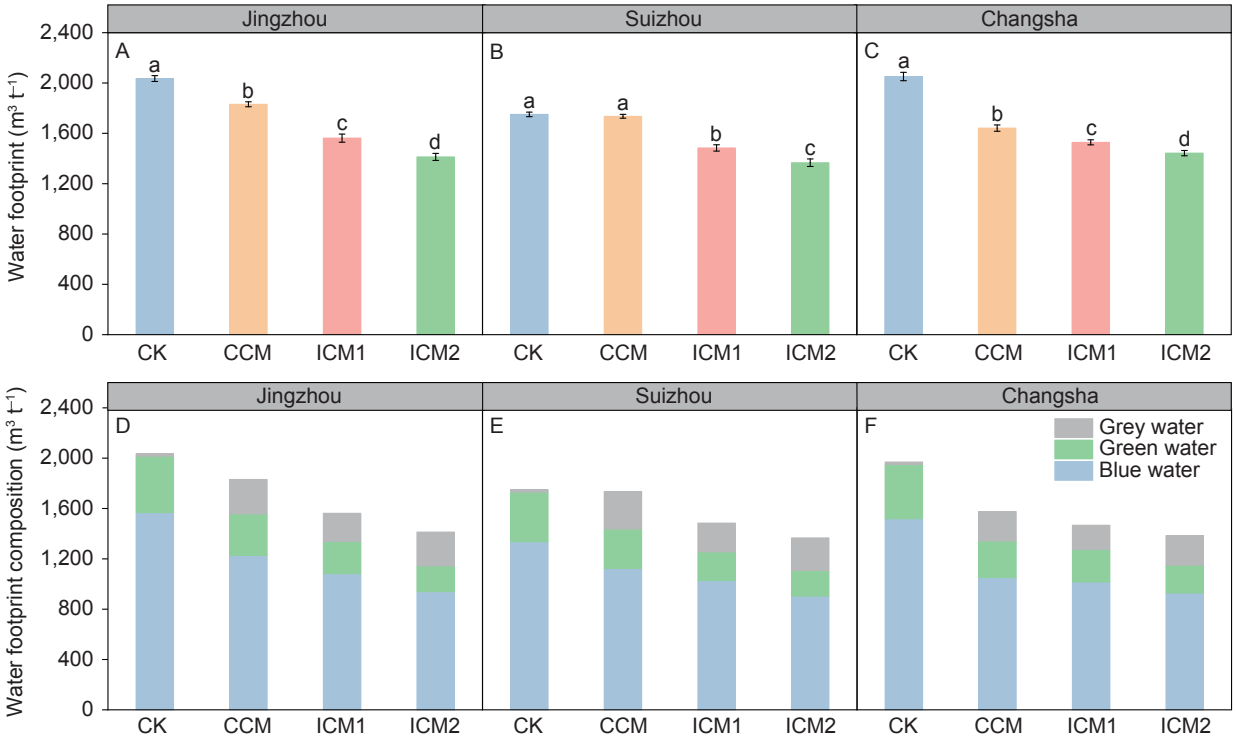


Fig. 7 Average water footprint values (A–C) and water footprint composition (D–F) across four crop management practices in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021. CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. Bars indicate SD (n=15, 9, and 6 for Jingzhou, Suizhou, and Changsha, respectively). Means sharing the same letter within each column are not significantly different according to LSD (0.05).

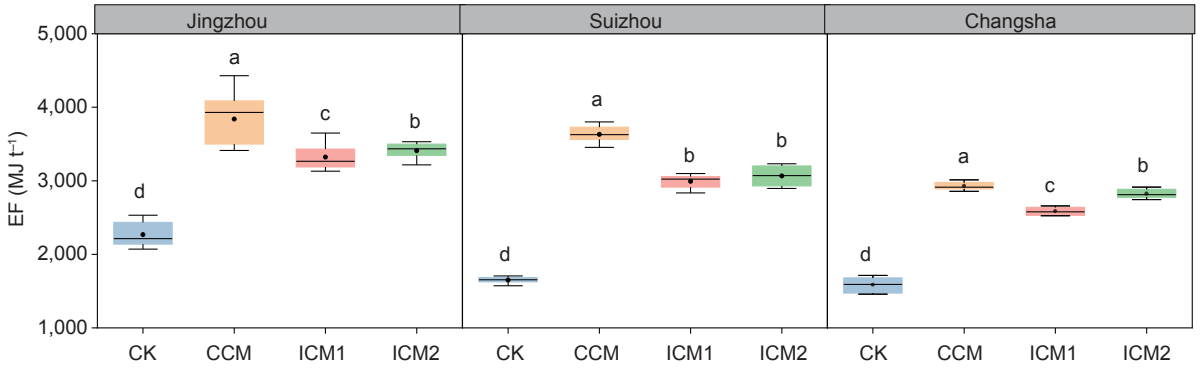


Fig. 8 Average values of energy footprint (EF) under four crop management practices in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021. CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. Bars indicate SD (n=15, 9, and 6 in Jingzhou, Suizhou, and Changsha, respectively). Means sharing the same letter do not differ significantly according to LSD (0.05).

ICM2 consistently achieved the highest sustainability score, followed by ICM1, with both exceeding 50 points. Notably, ICM2 demonstrated balanced performance across all indicators, with individual scores above 50, despite relatively lower energy and nitrogen scores. ICM1 also showed good balance in Suizhou and Changsha, with scores above 50, but performed less favorably in Jingzhou. Overall, both ICM2 and ICM1 outperformed CCM in terms of sustainability and balance, with ICM2 being particularly prominent.

4. Discussion

This study provides a comprehensive assessment of the dual impact of super rice on food security and environmental sustainability in China. While recognizing the critical role of super rice's high yield in addressing food security, we also examine potential environmental trade-offs associated with its production. Using a FCNWEP framework, we systematically evaluate greenhouse gas emissions, reactive nitrogen loss, water consumption, and energy use throughout the super rice production cycle. The results aim to inform decision-makers and guide future food production strategies, emphasizing

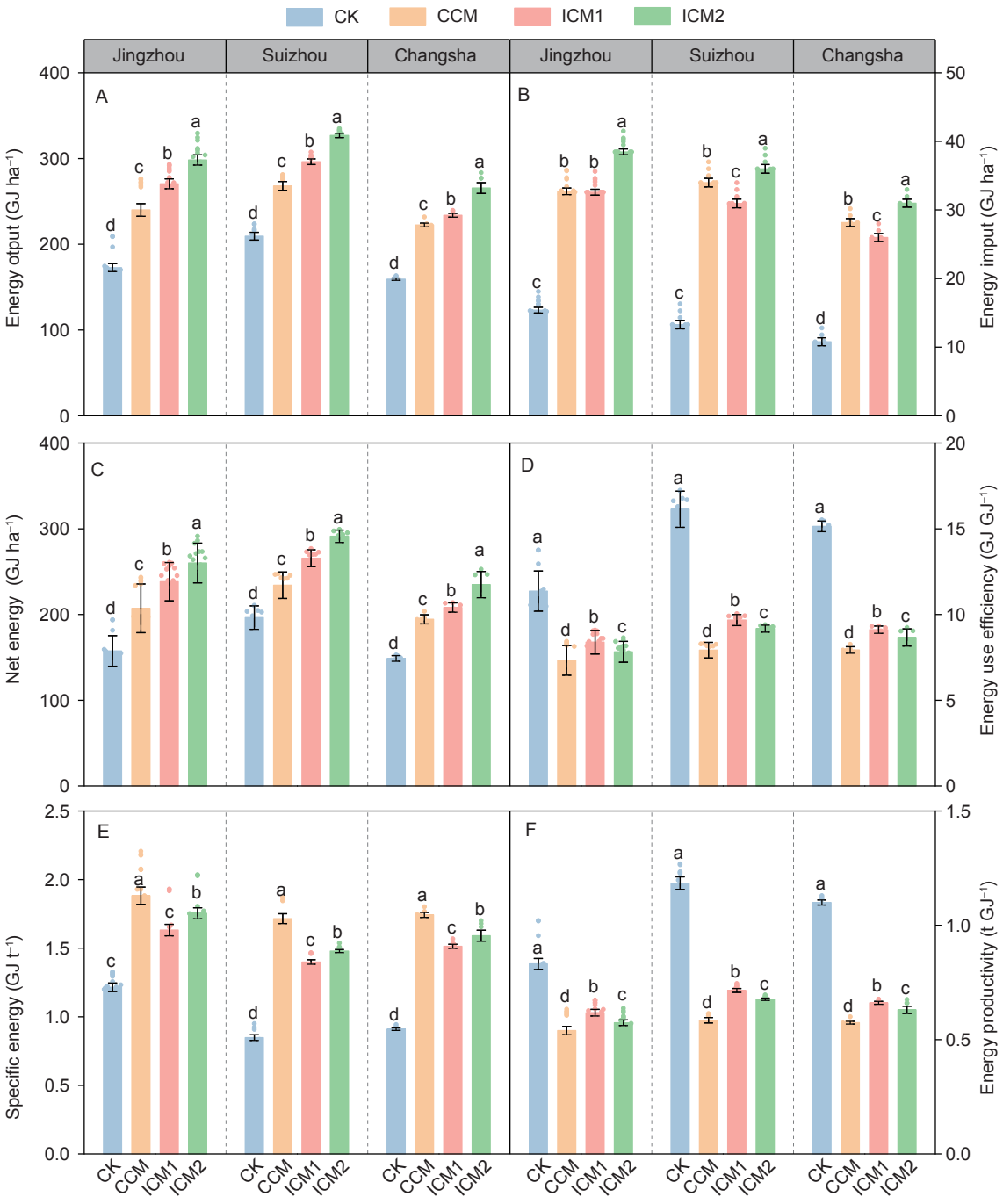


Fig. 9 Average values of energy output (A), energy input (B), net energy (C), energy use efficiency (D), specific energy (E), and energy productivity (F) under four crop managements in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021. Bars indicate SD (n=15, 9, and 6 in Jingzhou, Suizhou, and Changsha, respectively). No statistically significant differences were observed among means that share the same letter within each column based on LSD test at P=0.05.

the need to balance food security goals with environmental sustainability considerations.

4.1. Yield differences between four crop management in three sites

Both ICM1 and ICM2 demonstrated higher yields than CCM (Fig. 4). Previous studies have shown that a strategic reduction in nitrogen fertilizer application, combined with increased planting density, can mitigate yield losses by enhancing panicle formation (Lu *et al*. 2020; Deng *et al*. 2022a). Both ICM2 and ICM1 optimized nitrogen fertilizer application by increasing dosage during panicle initiation and flowering stages in the late growth phase — a strategy known to improve rice yields and nitrogen use efficiency (Qiao *et al*. 2022; Cai *et al*. 2023; Deng *et al*. 2023). In contrast, early-stage nitrogen application primarily influences tillering and spike density, whereas late-stage application has a greater effect on spikelets per panicle and seed-setting rates (Ma *et al*. 2021). Furthermore, research indicates that AWD

Table 3 Economic benefits of rice production under four crop management systems in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021

Site	Treatment[1)]	Income (USD ha^{-1})	Costs (USD ha^{-1})	Net profit (USD ha^{-1})	Gross margin (%)	Economic productivity
Jingzhou	CK	2,769 d	2,305 d	465 d	16 d	0.20 d
	CCM	3,539 c	2,799 c	740 c	20 c	0.26 c
	ICM1	4,005 b	2,795 b	1,209 b	30 b	0.43 b
	ICM2	4,607 a	3,014 a	1,593 a	35 a	0.53 a
Suizhou	CK	3,238 d	2,510 d	727 c	22 c	0.29 c
	CCM	3,839 c	3,116 b	724 c	19 d	0.23 d
	ICM1	4,215 b	3,066 c	1,149 b	27 b	0.37 b
	ICM2	4,789 a	3,270 a	1,519 a	32 a	0.46 a
Changsha	CK	2,744 c	2,284 d	460 d	16 c	0.20 c
	CCM	3,945 b	2,800 b	1,145 c	29 b	0.41 b
	ICM1	4,086 b	2,755 c	1,331 b	33 a	0.48 a
	ICM2	4,471 a	2,936 a	1,535 a	34 a	0.52 a

[1)] CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively.
Data presented are all mean values. No statistically significant differences were observed among means sharing the same letter within each column, according to the LSD test at P=0.05.

irrigation promotes the translocation of carbohydrates from vegetative organs to seeds, thereby increasing yield (Zhang *et al*. 2009). Notably, both ICM1 and ICM2 implemented AWD irrigation, which likely contributed to their elevated yields.

4.2. Production carbon footprint and nitrogen footprint

Fertilization and irrigation practices are key factors influencing rice yield and greenhouse gas emissions. In this study, ICM1, compared to CCM, demonstrated improved nitrogen use efficiency due to reduced nitrogen fertilizer input and increased planting density (Fig. 6-C and D). These practices effectively reduced nitrogen surplus, leading to lower N_2O emissions (Chen *et al*. 2014; Zhang *et al*. 2019). Conversely, ICM2, while improving nitrogen use efficiency, resulted in a notable increase in nitrogen surplus and subsequent N_2O emissions due to a substantial rise in total nitrogen applied. Research suggests that flooded rice paddies generally produce lower N_2O emissions, whereas alternate wetting and drying irrigation may create a trade-off between CH_4 and N_2O emissions (Johnson-Beebout *et al*. 2009; Riya *et al*. 2017; Verhoeven *et al*. 2018), consistent with our findings (Fig. 4-A).

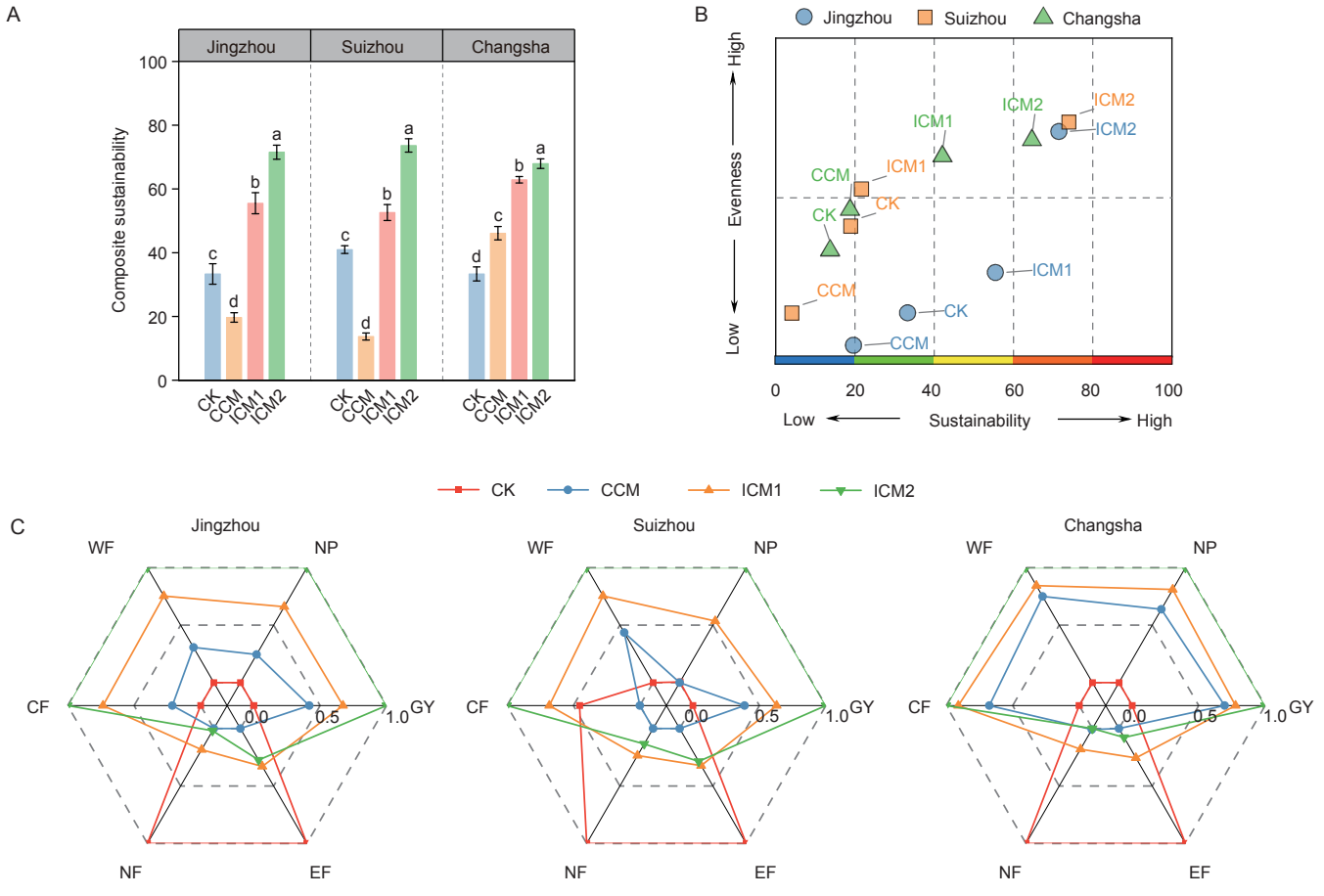


Fig. 10 Composite sustainability (A), the distribution of sustainability and evenness for all crop managements at each site (B), and performance across sustainability domains (C) under four crop managements in Jingzhou, Suizhou, and Changsha in Hubei and Hunan provinces, China from 2017 to 2021. Bars indicate SD (n=15, 9, and 6 in Jingzhou, Suizhou, and Changsha, respectively). CK, the control; CCM, conventional crop management; ICM1 and ICM2, integrated crop managements 1 and 2, respectively. GY, grain yield; CF, carbon footprint of production; NF, nitrogen footprint of production; WF, water footprint; EF, energy footprint; NP, net profit. No statistically significant differences were observed among means sharing the same letter within each column according to LSD (0.05).

Moreover, non-continuous irrigation has been shown to reduce global warming potential by up to 44% compared to continuous irrigation (Jiang *et al*. 2019). Additionally, both ICM1 and ICM2 exhibited significantly lower CF than CCM (Fig. 5), indicating that integrated crop management can simultaneously enhance yield and reduce CF.

In this study, NH_3 volatilization emerged as the primary source of nitrogen footprint, consistent with prior research (Leip *et al*. 2014). Clear differences in total nitrogen emissions across crop management practices and sites were evident (Fig. 6-A and B). These variations likely stem from a complex interplay of factors, including management practices, agricultural inputs (e.g., synthetic fertilizers), cultivation methods (e.g., irrigation), climate conditions (e.g., precipitation and temperature), and soil types. These factors collectively influence direct and indirect nitrogen oxide emissions and other nitrogen loss pathways. Nitrogen loss poses environmental and health risks; while these cannot be entirely eliminated, appropriate management practices can minimize their impact (Choudhury and Kennedy 2005). The fate of nitrogen in soil is governed by processes such as plant uptake, ammonia volatilization, denitrification, and surface or subsurface leaching.

4.3. Water footprint of rice production

Variations in water footprints among crop management practices were substantial, primarily driven by WF_{blue} (Fig. 7). The blue water footprint accounted for over 60% across all practices. Reducing blue water percolation in paddy fields can yield economic benefits by lowering water demand and supply costs, though environmental benefits may be limited (Chapagain and Hoekstra 2011).

AWD widely adopted in Asian countries such as China, the Philippines, Vietnam, and India, plays a significant role in conserving water and reducing emissions in rice cultivation (Lampayan *et al*. 2015). In this study, the water footprint of both ICM1 and ICM2 was significantly reduced, attributable to the implementation of AWD. WF_{green} does not impose significant environmental or economic burdens. Reductions in WF_{grey}, associated with leaching of fertilizers and pesticides, can be achieved through improved water use efficiency.

4.4. Energy footprint and energy flows of rice production

In this study, the higher energy output and net energy under ICM2 were primarily due to its elevated grain yield. For ICM1, in addition to high yield contributing to increased energy output, net energy was further enhanced by a 4.4% reduction in energy input compared to CCM. In terms of energy efficiency and productivity, CK showed the highest values, largely due to its substantially lower energy input, resulting in lower specific energy input per unit of yield.

Fertilizer use constitutes a major portion of energy input (Jin *et al.* 2023a), and ICM1, by reducing fertilizer application, improved its energy efficiency and productivity. Prior research indicates that using appropriately sized machinery, applying fertilizers based on soil needs, and following recommended crop management practices can enhance energy efficiency (Zhang *et al*. 2021). Yuan *et al*. (2022) found that direct seeding, compared to transplanting, has the potential to increase economic returns and energy output by reducing energy inputs and emissions. In this study, the reduction in energy footprint under ICM1 resulted not only from higher yield but also from decreased inputs of fertilizer, irrigation water, labor, and electricity. In contrast, ICM2, despite increased inputs of fertilizer, labor, and electricity, managed to reduce its energy footprint, primarily due to a substantial yield increase and reduced irrigation water use.

4.5. Implications and limitations of this study

This study is based on the FCNWEP system framework, highlighting interconnections among these domains. This approach aids in optimizing agricultural management strategies for sustainable development by identifying synergies and trade-offs to inform decision-making. First, reducing reliance on nitrogen fertilizers should be a priority for future research, as they are the most energy-intensive input and contribute to greenhouse gas emissions and water eutrophication. Second, given the increasing frequency of extreme weather events such as heatwaves and droughts, along with rapid population growth (Liu *et al*. 2023), water scarcity is a growing concern. Therefore, further investigation into water-saving irrigation techniques, such as alternate wetting and drying, is urgently needed. A multi-objective integrated crop management approach that considers interactions within the FCNWEP system and the impacts of integrated practices offers a valuable framework for designing cropping systems that balance economic and ecological outcomes. Scientifically and rationally improving resource utilization in agriculture presents both opportunities and challenges for sustainable food production.

However, this study also has limitations, suggesting directions for future research. First, only one rice variety, YLY900, was used. Different crop types or varieties may differ in growth, development, and disease resistance, limiting the generalizability of the findings. Second, data for CF, NF, EF, and WF were calculated rather than empirically measured. The use of different calculation methods or models could yield varying footprint estimates, potentially affecting comparability across studies. Third, soil organic carbon is a critical carbon reservoir, important for assessing overall carbon footprint and for climate change mitigation and soil quality improvement. In this study, changes in soil organic carbon content were not quantified, which may introduce bias in carbon footprint assessment and limit the accuracy of carbon cycling evaluation.

5. Conclusion

This study employed a comprehensive FCNWEP framework to evaluate different crop management practices. Results show that integrated crop management significantly enhances rice yield, resource use efficiency, and economic profits compared to conventional practices. Importantly, these approaches concurrently reduce carbon footprint, contributing to greater agricultural sustainability. They also lead to reductions in nitrogen, water, and energy footprints, improving resource efficiency and reducing environmental pressure.

In conclusion, integrated crop management practices significantly enhanced rice yield, resource use efficiency, and overall sustainability compared with conventional practices. These approaches concurrently reduced carbon, nitrogen, water, and energy footprints, demonstrating strong potential for eco-efficient rice production systems. These findings provide a scientific basis for promoting eco-efficient rice production and inform policy decisions toward sustainable agricultural development in China.

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Declaration of competing interest

The authors declare that they have no conflict of interest

Declaration of generative AI and AI-assisted technologies in the writing process

The authors declare that they did not use AI in the preparation and writing of this manuscript.

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