

Review

Combined Effects of Water-Saving Irrigation and Biochar Application on Greenhouse Gas Emissions from Paddy Fields: A Review

Chaobiao Meng ^{1,*} , Liyan Zhang ², Yifan Xie ¹, Yumeng Gao ¹, Jinghao He ¹, Keren Yang ¹, Feiyan Xu ³, Weihai Wang ^{1,*}, Yanbing Chi ¹ and Si Chen ¹

¹ School of Hydraulic Engineering, Zhejiang University of Water Resources and Electric Power, Hangzhou 310018, China; xieyf@stu.zuwe.edu.cn (Y.X.); gaoym@stu.zuwe.edu.cn (Y.G.); hejh@stu.zuwe.edu.cn (J.H.); yangkr@stu.zuwe.edu.cn (K.Y.); chiyb@zuwe.edu.cn (Y.C.); chensi6@zuwe.edu.cn (S.C.)

² Jiaxing HangJiaHu South Drainage Project Management Service Center, Jiaxing 314000, China; zly13567397630@sina.com

³ River, Lake and Rural Water Conservancy Management Service Center of Haining City, Haining 314400, China; xfyhaining@163.com

* Correspondence: mengcb@zuwe.edu.cn (C.M.); wangweiha@zuwe.edu.cn (W.W.)

Abstract

Rice paddies are a major agricultural source of greenhouse gas emissions. However, balancing emission reduction with food security remains a core challenge for climate-smart rice production management. This review synthesizes current evidence on water-saving irrigation (WSI), biochar, and their combined effects on methane (CH₄), nitrous oxide (N₂O), global warming potential (GWP), and greenhouse gas intensity (GHGI) in paddy fields, emphasizing interaction mechanisms and environmental and management constraints. WSI generally suppresses CH₄ emissions but can increase N₂O emissions during drying–rewetting transitions. Mean reductions of approximately 43–52% in CH₄ emissions and 37–47% in GWP have been reported in studies of alternate wetting and drying relative to flooded controls. Biochar can modify these responses through changes in soil aeration, pH, carbon availability, and nutrient retention. Under WSI, reported N₂O decreases following biochar addition ranging from 19.3% to 62% relative to no-biochar controls under the same irrigation regime, although these decreases are not uniformly significant, and nonsignificant responses also occur. Additional CH₄ mitigation is likewise context-dependent, with increases reported under some conditions. When CH₄ reductions outweigh N₂O increases in CO₂-equivalent terms while yield remains stable, combined management can lower field-emission GWP and GHGI. A net benefit across the full production system additionally depends on carbon storage and upstream emissions. Excessive drying, poorly coordinated nitrogen inputs and unsuitable biochar–soil combinations can weaken these benefits. Uncertain aging effects, limited long-term multi-site evidence and inconsistent accounting boundaries constrain the assessment of persistence and transferability. Integrated WSI–biochar management therefore offers conditional mitigation potential, requiring locally evaluated coordination of water, nitrogen, and biochar inputs rather than uniform prescriptions.

Keywords: paddy field; methane; nitrous oxide; alternate wetting and drying; biochar; GWP; GHGI



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1. Introduction

Rice (*Oryza sativa* L.) is one of the world's most important staple crops, meeting the dietary needs of more than half of the global population, and underpins food security across much of Asia [1]. Globally, rice is predominantly cultivated in Asia, which accounts for approximately 90% of global production, with significant areas also in Africa, the Americas, and Europe. Rice crops occupy more than 10% of the world's arable land, highlighting its massive footprint on agricultural water use and greenhouse gas inventories [1]. Conventional paddy cultivation, particularly prolonged flooding, contributes substantially to methane (CH_4) emissions. Methane has a 100-year global warming potential, approximately 27 times that of carbon dioxide (CO_2) [2,3]. Such climate impacts are now being worsened by an intensifying water crisis. Urban expansion and climatic change are bringing tighter competition for freshwater between sectors (agriculture, homes, and nature), which challenges the old assumption of continual flooding as both resource- and climate-friendly [2,3]. These pressures motivate rice-management practices that conserve water while limiting greenhouse gas emissions and maintaining crop production.

Water-saving irrigation (WSI), including alternate wetting and drying (AWD), controlled irrigation (CI), and mid-season drainage (MD), is widely studied as an alternative to continuous flooding [4]. By interrupting prolonged anaerobic conditions, WSI can reduce CH_4 emissions, but drying and rewetting may increase N_2O emissions when mineral nitrogen is available [5,6]. These opposing changes do not automatically cancel each other: their climate effects must be compared in CO_2 -equivalent terms. Area-based GWP and yield-scaled GHGI provide complementary measures of this trade-off. Reported benefits depend on the actual drainage schedule, nitrogen management, and flooded or conventional comparator [7,8].

Biochar is a carbon-rich, highly porous material produced via biomass pyrolysis under oxygen-limited or anoxic conditions. Its GHG-relevant properties—including pore network structure, surface pH, ash alkalinity, cation exchange capacity (CEC), and carbon recalcitrance—vary substantially depending on feedstock biochemistry and thermochemical conversion parameters, rendering biochar a highly heterogeneous soil amendment rather than a uniform input [9,10]. Common types of biochar used in agriculture are derived from crop residues (e.g., rice straw, wheat straw, and maize stover), wood biomass, and animal manures, typically produced at pyrolysis temperatures between 300 °C and 700 °C [9]. Its agricultural application is actively researched and increasingly adopted in countries such as China, the United States, Australia, and several European nations, where it is utilized for soil amelioration and carbon sequestration [9,11]. By modifying soil aeration, pH, nutrient retention, and carbon availability, biochar may complement irrigation management [9,11–13]. These changes can influence both CH_4 production and oxidation and the balance between N_2O production and consumption. However, feedstock, production conditions, the application rate, and the receiving soil affect the direction and magnitude of the response [14]. Biochar should therefore be evaluated as a context-dependent amendment rather than assumed to resolve the CH_4 – N_2O trade-off in every paddy system.

Among real-field studies, biochar applications are evaluated with a variety of experimental setups. These may be limited to short-term single-season trials—i.e., of immediate yield or emission relevance—or they may cover long-term multi-year observation of biochar aging and continuing influence; multi-site networks are used to check performance under several climatic and soil configurations [15–17]. Field-based analysis varies in breadth: it may deal only with plot-level data (on yield or soil) or go on to consider life-cycle impacts: production-related emissions and long-term carbon storage in soil [12,14].

Even with the rapidly growing literature, three persistent gaps still limit the reliability of recommendations. First, most current studies assess WSI or biochar separately, whereas

climate-smart practice increasingly calls for their combined use in management. Second, experimental time frames are often too short to determine reliably whether an effect persists through many wet–dry cycles or microbial adaptation or biochar aging. Third, large regional heterogeneity—among climates, soils, and modes of farming—complicates attempts to apply globally derived effect sizes to localized decision making [15–17]. The urgency of mitigating agricultural greenhouse gas emissions while ensuring food security necessitates a comprehensive synthesis of how water-saving irrigation and biochar interact. While several recent studies have begun to evaluate their combined application [10,12,13,16–19], there remains a critical need for an article aimed at compiling, synthesizing, and specifying the operational parameters and resulting outcomes. This review addresses this gap by evaluating the underlying thermodynamic, enzymatic, and kinetic mechanisms of combined WSI and biochar application. By doing so, this synthesis establishes a foundation for future research and uncovers new avenues of opportunity for optimizing greenhouse gas mitigation in paddy fields.

2. Methodology

2.1. Literature Search Strategy

To ensure a comprehensive and systematic review of the literature, a structured search strategy was employed. The literature search was conducted using the Web of Science core database to ensure the inclusion of high-impact, rigorously peer-reviewed studies and maintain the highest academic standards. The search covered publications from 2010 to September 2026, reflecting the period of most intensive research on this topic. The search algorithm utilized a combination of keywords and Boolean operators. The primary search string was (“water-saving irrigation” OR “alternate wetting and drying” OR “controlled irrigation” OR “mid-season drainage”) AND (“biochar” OR “black carbon”) AND (“greenhouse gas*” OR “methane” OR “CH₄” OR “nitrous oxide” OR “N₂O” OR “global warming potential” OR “GWP” OR “greenhouse gas intensity” OR “GHGI”) AND (“paddy” OR “rice”).

This highly specific Boolean logic returned 62 records. While this number may appear restrictive for a 16-year period, it accurately reflects the scarcity of studies that simultaneously evaluate both interventions (WSI and biochar) and their combined effects on greenhouse gases in paddy fields, rather than evaluating them in isolation. After the second phase of screening based on the selection criteria shown in Section 2.2, 54 papers were retained for final analysis. Because the reported search string requires both irrigation and biochar terms, it intentionally filters out the vast body of literature that addresses either practice alone, thereby ensuring the synthesis remains tightly focused on their combined interactive effects.

2.2. Selection Criteria

The selection of literary sources was based on the following inclusion criteria: (1) peer-reviewed research articles, reviews, and meta-analyses published in English; (2) studies conducted in field or greenhouse conditions involving rice cultivation; (3) studies reporting quantitative data on CH₄ and/or N₂O emissions, GWP, or GHGI; and (4) studies evaluating combined effects of WSI and biochar. Exclusion criteria included (1) studies lacking a continuous flooding control or a non-biochar control; (2) purely modeling studies without empirical validation; and (3) conference abstracts, short communications, or non-peer-reviewed reports. After removing duplicates, the titles and abstracts were screened, followed by a full-text assessment to determine final eligibility.

2.3. Data Extraction and Synthesis

For each study considered in the synthesis, extracted information comprises site characteristics (climate, soil texture, initial soil organic carbon, and pH), experimental design (water regime, biochar feedstock, production temperature, and application rate), nitrogen management (fertilizer type, rate, and timing), and outcomes (CH_4 , N_2O , grain yield, GWP, and GHGI).

Quantitative comparisons are retained with their actual treatment, control, observation period, and study type. Irrigation effects relative to flooded controls, biochar effects relative to no biochar within the same irrigation regime, and combined-treatment effects relative to a specified baseline are reported separately. Percentage change is defined as $100 \times (\text{treatment} - \text{control}) / \text{control}$; negative and positive values indicate decreases and increases, respectively. The numerical examples below retain the changes reported by the source studies rather than reconstructing unavailable treatment means.

2.4. Measurement Quantities and Climate Indicators

Gas concentration, flux, and cumulative emissions represent different levels of greenhouse gas measurements. Gas concentration is generally reported as a mole fraction (e.g., ppm or ppb), whereas gas flux represents the rate of gas exchange per unit soil area and time. Cumulative emissions are obtained by integrating fluxes over the observation period and are commonly expressed as kg gas ha^{-1} . In this review, greenhouse gas mitigation is primarily evaluated using cumulative CH_4 and N_2O emissions and their corresponding climate indicators.

Global warming potential (GWP) is used to express the warming impact of different greenhouse gases on a common CO_2 -equivalent basis over a specified time horizon. For field emissions of CH_4 and N_2O , area-based GWP can be calculated as:

$$\text{GWP} = E(\text{CH}_4) \times \text{CF}(\text{CH}_4) + E(\text{N}_2\text{O}) \times \text{CF}(\text{N}_2\text{O}),$$

where E represents cumulative CH_4 or N_2O emissions (kg gas ha^{-1}), and CF represents the corresponding characterization factor ($\text{kg CO}_2\text{-equivalent kg}^{-1} \text{ gas}$). Thus, GWP is expressed as $\text{kg CO}_2\text{-equivalent ha}^{-1}$.

Greenhouse gas emission intensity (GHGI) further relates greenhouse gas emissions to crop production and is calculated as:

$$\text{GHGI} = \text{GWP}/Y,$$

where Y is grain yield (kg ha^{-1}). GHGI is therefore expressed as $\text{kg CO}_2\text{-equivalent kg}^{-1}$ grain. GWP reflects greenhouse gas emissions on a land-area basis, whereas GHGI reflects emissions per unit of grain production. These two indicators provide complementary perspectives for evaluating the climate implications of rice production.

3. Effects of Water-Saving Irrigation on Greenhouse Gas Emissions

Among the principal working forms of WSI—that is, AWD, CI, and MD—each one aims essentially to minimize non-productive water loss, but it produces unique soil moisture histories and characteristic redox conditions [20–34]. The thresholds of drying for each method, the frequency of drainage, and the reflooding rules do not occur at random but are useful in determining gradients of biogeochemistry relevant to CH_4 suppression, N_2O pulses, and plant yields (Table 1).

Table 1. Quantitative examples for WSI practices.

WSI Type	Core Hydrological Feature	CH ₄ Change	N ₂ O Change	Yield Context
Alternate wetting and drying irrigation [4,5,24,32]	Cyclic wetting–drying with threshold re-irrigation	−71—−43.23% ¹	+42.59–44.0%	0–+9%
Controlled irrigation [22,23]	Moisture maintained in moderate target range	−58—30%	+85–137%	−6–+8.6%
Mid-season drainage [7,27,28,33,34]	Planned seasonal drainage window	−63.7–0%	+13.4–137%	0–+6%
Shallow intermittent flooding [29]	Thin water layer + frequent drainage	−45—34%	+27–50%	0–+4.7%

¹ Percentages denote reported changes relative to each stated control, unless explicitly labeled as model predictions. Negative/positive values indicate decreases/increases; reduction ranges are written as positive magnitudes where specified.

3.1. GHG Emission Outcomes and the CH₄–N₂O Trade-Off

CH₄ suppression is a recurring response to water-saving irrigation, consistent with the interruption of anaerobic conditions and increased opportunities for methane oxidation (Figure 1A) [5,7]. Two global AWD meta-analyses reported reductions of 43.23% [4] and 51.6% [5] relative to flooded controls; a broader synthesis, including AWD, CI, and other WSI practices, reported a 71.0% reduction [32]. Field observations under AWD, single mid-season drainage, shallow irrigation, and intermittent irrigation support the same direction of response [7,8,24,29]. Together, these findings indicate that shortening the duration of reducing conditions can lower CH₄ across different irrigation schedules, although the separate pooled estimates describe their respective evidence bases rather than the endpoints of an expected field-response interval. Additional CH₄ reductions are less consistent when the comparator already incorporates drainage. A Japanese nine-site comparison summarized in a review reported a 30% reduction from prolonging conventional MD [27], whereas a separate two-year field study found no significant annual CH₄ difference [28]. This contrast concerns the marginal benefit of extending drainage, rather than introducing drainage into continuously flooded rice. Similarly, the 30–58% reduction observed under CI in a pot experiment supports the aeration mechanism but does not establish its magnitude under field conditions [26]. The direction of response is therefore informative only in relation to the initial water regime and the setting in which it was measured.

However, these same redox transformations also accelerate the production of N₂O. The aeration intervals promote the oxidation of ammonium to nitrate (NH₄⁺ → NO₃[−]), and the subsequent rewetting events trigger denitrification, resulting in incomplete nitrate reduction and intermittent N₂O pulses [4,5,8]. For instance, Li et al. [4] and Zhao et al. [5] reported pooled N₂O increases of 42.59% and 44.0% under AWD, respectively. These pooled effects are consistent in direction with field increases under single MD, SI, and II [7,29] and the broader WSI synthesis [32]. Greater increases have also been observed, including an average of 210% under AWD in a two-year field trial [35] and 85–137% under CI in a pot experiment [26]. These results are consistent with enhanced nitrification–denitrification during drying and rewetting when mineral N is available. However, N₂O can also decrease or show no significant response: reductions of 12–70% occurred under AWD relative to a conventional control drained during fertilizer applications [8], while annual differences under prolonged versus conventional MD were nonsignificant [28]. Thus, N₂O increases are not an inevitable consequence of every water-saving schedule; fertilizer–drainage coordination and the control emission level help determine both the direction and relative magnitude of the response.

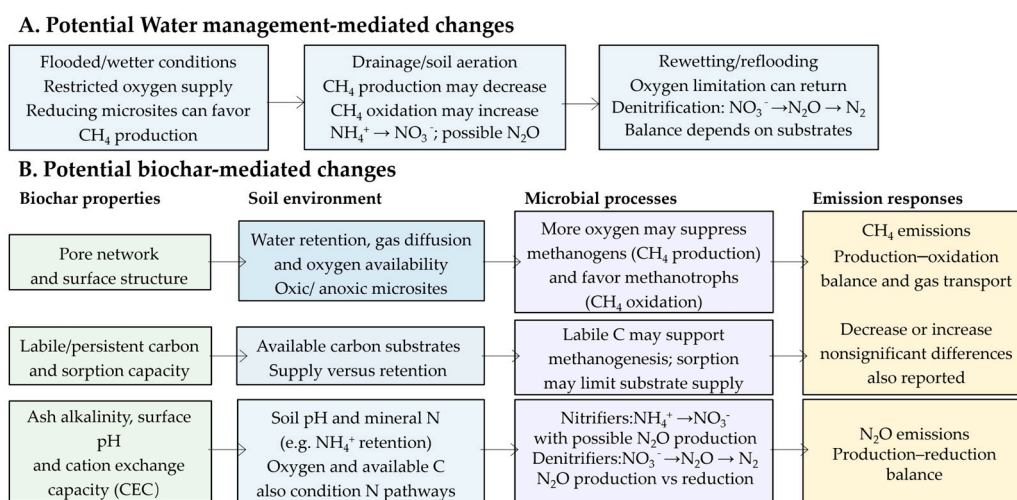


Figure 1. Conceptual pathways linking WSI, biochar and paddy greenhouse gas outcomes. WSI changes soil moisture and redox conditions, while biochar can modify carbon availability and nitrogen transformations. (A) Qualitative drying–rewetting sequence and possible carbon and nitrogen pathways. Box order indicates a process sequence, not measured time, Eh, or emission magnitude; N₂O can be produced during both nitrification and denitrification and consumed during further reduction to N₂. (B) Potential biochar-mediated changes, including competing carbon-supply, and retention effects; their net expression depends on material and soil conditions. Arrows indicate possible process links, not effect sizes or inevitable mitigation. Process interpretation draws on [4,5,9,12–14,18,20,34].

Despite this gas trade-off, reductions in integrated climate indicators recur across the evidence. Li et al. [4] reported GWP and GHGI reductions of 36.84% and 38.57%, respectively, with a yield increase of 1.52%; Zhao et al. [5] reported a 46.9% reduction in CH₄ + N₂O GWP. For the broader WSI category, Xing et al. [32] reported GWP and GHGI reductions of 54.7% and 51.7%, respectively, with overall yield stability. From a CO₂-equivalent perspective, as long as the CH₄ emission reduction effect exceeds the accompanying increase in N₂O emissions, a well-optimized WSI regime can still achieve net GWP mitigation.

3.2. Integrated Performance and Key Outstanding Gaps

The quantitative evidence supports the potential for WSI to lower GWP and GHGI but does not establish a universal response for all drainage schedules (Table 1). The distinction between practice-specific pooled effects and individual study outcomes is essential for interpreting both benefits and risks [4,5,32]. Four outstanding issues motivate the combined-management assessment in Section 5.

First, N₂O accounting remains structurally incomplete; specifically, research reliant on low-frequency (e.g., weekly) static chamber measurements frequently overlooks transient, high-intensity post-drainage emission pulses. This results in a systematic underestimation of GWP that artificially inflates apparent mitigation efficacy [5,8]. Second, threshold context-dependency represents a major bottleneck. Optimal re-irrigation thresholds are highly sensitive to soil texture, organic carbon content, and local climate; yet, a broadly validated threshold directory for diverse paddy environments is still lacking [4,5,29]. Third, the long-term performance of WSI under multi-year dry–wet cycles is poorly constrained, with the literature dominated by short-term (one- or two-season) studies that offer little insight into cumulative microbial community adaptation and soil substrate depletion trajectories [24,25,29]. Fourth, the synchronization of nitrogen (N) fertilizer with water management is frequently overlooked. Given that the N₂O-mediated feedback is highly sensitive to the form and timing of N applications, the lack of factorially crossed water-

and-nitrogen treatments in most WSI trials obscures vital interactive effects [6,30]. These gaps represent the exact biogeochemical leverage points where biochar co-application (detailed in Section 5) can buffer soil redox oscillations and modulate nitrogen transformation pathways, establishing the core scientific rationale for the integrated management strategy evaluated in Section 4.

4. Effects of Biochar on Greenhouse Gas Emissions

Biochar is a carbon-rich, highly porous material produced via biomass pyrolysis under oxygen-limited or anoxic conditions. Its GHG-relevant properties—including pore network structure, surface pH, ash alkalinity, cation exchange capacity (CEC), and carbon recalcitrance—vary substantially depending on feedstock biochemistry and thermochemical conversion parameters, rendering biochar a highly heterogeneous soil amendment rather than a uniform input (Figure 1B) [9,10].

Higher pyrolysis temperatures generally yield highly aromatic, condensed, and chemically recalcitrant carbon structures with superior structural stability but highly variable nutrient profiles; conversely, lower thermal treatment yields biochars characterized by larger fractions of functionalized, labile carbon complexes [9,14]. Because these physicochemical properties co-vary intimately with initial feedstock chemistry, univariate generalizations predicated solely on pyrolysis temperature remain oversimplifications; consequently, site-specific “property-soil matching” is an absolute prerequisite for achieving predictable and beneficial agro-environmental and GHG outcomes [9–11].

4.1. GHG Emission Responses: Mechanisms and Magnitude

Biochar-induced CH₄ responses include both decreases and increases, reflecting competing effects on gas diffusivity, carbon availability, pH, and microbial activity [9,12,13,20]. Under CI, lower CH₄ emissions were reported following rice-straw biochar addition in initial and later observations on the same Taihu experimental platform [13,36]. Dose-specific reductions were 29.7% and 15.6% at 20 and 40 t ha^{−1} in the initial study [36] and 22.76% and 37.52% at the corresponding doses in the later gas season [13]. These observations support the potential for mitigation within CI, but their differing dose patterns and shared site preclude a general dose–response relationship or independent geographical replication. Contrarily, a 38% increase associated with rice-straw biochar was reported across the studied AWD and permanently flooded regimes relative to no organic amendment [37]. The coexistence of decreased and increased emissions indicates that improved aeration or methane oxidation need not outweigh the influence of available carbon and other amendment-induced changes at every site.

For N₂O, several field studies have reported that biochar addition reduces N₂O emissions under water-saving irrigation, with reported reductions ranging from 19.3% to 62% relative to no-biochar controls under the same irrigation regime [12,13,35,38]. This descriptive range summarizes the reductions reported in these cited field studies rather than a pooled estimate or a universal mitigation rate. Such mitigation may be associated with changes in soil pH, nutrient retention, and microbial N transformations that favor N₂O consumption over production [9,20]. However, the response is not consistent across studies: Pandey et al. (2014) [37] found no significant change in cumulative N₂O emissions following rice-straw biochar addition relative to the corresponding no-organic-amendment controls under either AWD or permanent flooding. These contrasting findings indicate that the magnitude and statistical significance of biochar-induced N₂O mitigation depend on biochar properties, soil conditions, and water and nitrogen management.

4.2. Key Uncertainties: Aging, Context Dependency, and Evidence Gaps

Four uncertainties limit the possibility of translating site-specific biochar trials into scalable, field-oriented management rules.

First, biochar aging dynamically modifies its physicochemical properties. Specifically, processes such as surface oxidation, organo-mineral coating development, and pore blockage progressively affect, with time, gas diffusivity, redox reactivity, and nutrient retention capacities [17,20]. Because most of paddy biochar trials reported so far deal only with one or two growing seasons, they do not fully clarify how these gradual functional changes influence CH₄ and N₂O across several years' deployments. For example, Yang et al. [39] found no significant GWP difference between CI with and without biochar in the first rice season, whereas 20 and 40 t ha⁻¹ reduced CH₄ + N₂O GWP by 35.7% and 21.5%, respectively, in the second season. Moreover, percentages depend on control emission levels [40].

Second, mismatched combinations of properties and soils help explain many of the null or inconsistent findings in published work. For instance, a 22.9% CH₄ reduction (corresponding to an average regional value under biochar amendment [9,11,14]) of emissions from a high-emission paddy may yield an absolute climate benefit of much greater magnitude than one derived from a higher numerical percentage of reduction achieved at a low-emission baseline, so that relative effect sizes need not be assumed valid across sites without detailed baseline analysis [9,32].

Third, the economic and practical viability of biochar application at the farm scale remains largely underexplored. High rates (e.g., 20–40 t ha⁻¹) recommended in experimental trials are unlikely to be cost effective under most farming conditions unless there is an established local feedstock source and long-term performance data justifying the upfront capital [12,36,39].

Fourth, the effect of biochar application is not uniform but is very much dependent upon the original soil properties, especially pH and granulometric features (i.e., soil texture). Among other things, biochar proves particularly helpful in acidic soils because of its capacity to buffer pH and therefore promote N₂O reductase activity (with reduced N₂O emission), while such effects—when applied to highly alkaline soils—may be minimal or even harmful [9,11,14]. Soil texture, too, influences how biochar behaves: porous biochar added to coarse, sandy soils helps with water and nutrient holding, whereas its use for fine, clay-like soils tends to improve aeration and to lower bulk density, thus helping to balance methanogenesis and methane oxidation [9,32]. Therefore, biochar should not be treated as a one-size-fits-all solution but rather as a context-dependent technology.

Taken together, biochar can improve the performance of WSI, but reductions, non-significant differences and increases all occur. The combined-management assessment in Section 5 therefore distinguishes the observed response from a proposed mechanism and from a statistically established interaction.

5. Combined WSI + Biochar Effects: Evidence, Mechanisms, and Boundary Conditions

5.1. Rationale for the Combined Strategy

WSI and biochar influence overlapping soil processes through different management interventions. The temporal basis for combining them is better understood as shorter drying–rewetting events superimposed on crop-season changes in water management, rather than as a fixed season-to-week redox cycle. Flooding for crop establishment or flowering, intervening drainage and re-irrigation, and pre-harvest drying determine when the soil is exposed to contrasting oxygen conditions [4,5,18]. For example, successive irrigations in one AWD field schedule were approximately 10–11 days apart before a flooded

flowering stage [37]. This illustrates management timing, not a measured period of soil redox-potential (Eh) oscillation: rainfall, soil water retention, depth, and biological oxygen demand influence how rapidly and how far the soil responds [4,5].

These hydrological transitions affect CH_4 through the balance between production and oxidation. Under flooding, restricted oxygen replenishment and biological oxygen consumption favor reducing conditions and, where suitable carbon substrates are available, methanogenesis. Drainage increases aeration in the soil zones reached by air, tending to raise Eh, inhibit methanogenesis, and favor aerobic CH_4 oxidation [4,5]. Rewetting restricts oxygen supply again and can permit methanogenesis to resume as reducing conditions develop but does not necessarily restore the pre-drainage CH_4 flux. Carbon availability and the preceding aerated period also matter: low CH_4 fluxes persisted during the flooded flowering stage following AWD [37], consistent with a carry-over effect of the earlier drying periods rather than an immediate reversal controlled by water level alone. Thus, the duration and sequence of wet and dry conditions are more informative than assuming that every re-irrigation initiates an identical redox and emission cycle.

The same sequence can create a CH_4 – N_2O trade-off because nitrogen transformations respond differently to oxygen availability. During drainage, nitrification can convert available NH_4^+ to NO_3^- and produce N_2O ; denitrification can occur concurrently in oxygen-limited microsites. Following rewetting, the combination of restricted oxygen supply, available NO_3^- and degradable carbon can favor denitrification, during which N_2O may either escape or be further reduced to N_2 [4,38]. A transition therefore does not invariably generate an N_2O pulse: its outcome depends on the balance between production, consumption, and gas transport. Fertilizer application can increase the mineral-N supply during these transitions, whereas later crop uptake and substrate depletion can limit it [4,12,37]. Observed N_2O increases after topdressing and low emissions during later flooded growth support this timing dependence [37], without resolving the separate contributions of nitrification and denitrification from flux measurements alone.

Biochar can modify the soil conditions through which these events act, providing a rationale for combining the practices. Changes in pore structure may alter aeration, while pH, nutrient retention, and carbon availability can affect the substrates and microbial processes involved [12,13,36]. Observed differences in microbial community composition provide additional evidence of amendment–soil associations, although these do not establish the rates of the proposed processes [13]. Where amendment-induced changes favor CH_4 oxidation or N_2O reduction to N_2 , biochar may complement the CH_4 benefit of drainage and moderate its N_2O cost [12,38]. Conversely, added available carbon can support CH_4 production under reducing conditions, and improved nutrient retention does not guarantee lower N_2O throughout subsequent wetting–drying events. The contrasting CH_4 responses and the coexistence of N_2O reductions and nonsignificant responses under biochar addition [12,37] therefore support conditional complementarity rather than a uniformly beneficial mechanism.

5.2. Coupled Interaction Mechanisms

Water management and biochar can jointly influence the distribution of aerobic and anaerobic microsites, but drainage-induced changes in bulk-soil Eh do not establish uniform changes throughout the soil profile. Biochar properties may affect pore structure, carbon availability, and microbial activity, with consequences for both methanogenesis and methane oxidation [9,12,13,18]. The available emission comparisons do not establish that biochar spatially homogenizes Eh or that this process produces suppression beyond additivity. For example, 20 and 40 t ha^{−1} rice-straw biochar reduced CH_4 by 29.7% and 15.6%, respectively, relative to CI without biochar [36]. These values quantify an additional amendment effect

within CI; they neither measure the spatial distribution of Eh nor demonstrate superiority to both individual practices. The 75–89% reduction for AWD with acid-modified biochar relative to CF without biochar [41] instead describes the whole management package and cannot be used to infer the same mechanism.

With regard to the nitrogen axis, the main environmental liability of WSI is the episodic N_2O pulse brought about every drainage–rewetting cycle, in which nitrate (NO_3^-) accumulated under aerobic conditions undergoes rapid denitrification upon reflooding. Biochar can act as a buffer along this pathway: (i) adsorbing ammonium (NH_4^+) to moderate nitrification during the dry phase to control nitrification; (ii) raising soil pH so as to favor complete denitrification, which lowers the $\text{N}_2\text{O}:\text{N}_2$ product ratio; and (iii) making soil moisture more uniform to prevent abrupt shifts from aerobic to anaerobic states [9,12,20]. This buffering effect is, nonetheless, very sensitive to nitrogen inputs; if massive single fertilizer doses come at the drainage events, the transient NO_3^- supply overwhelms the adsorption capacity of the biochar matrix, rendering N_2O mitigation negligible.

From a thermodynamic perspective, the combined application of WSI and biochar alters the free energy yield of terminal electron-accepting processes [9,13]. Biochar's electron-donating and -accepting capacities can buffer redox fluctuations during WSI cycles, potentially shifting the thermodynamic favorability away from methanogenesis toward more energetically favorable processes like iron or nitrate reduction [19,26]. Enzymatically, biochar can influence the activity of key enzymes such as methanogenesis-related methyl-coenzyme M reductase (*mcrA*) and denitrification-related reductases (e.g., *nirK*, *nirS*, and *nosZ*) by altering soil pH, providing surface area for microbial colonization, and binding inhibitory compounds [13,19]. Kinetically, the porous structure of biochar can alter the diffusion rates of oxygen and substrates, thereby modifying the kinetics of methane oxidation and denitrification [9,26]. However, the exact kinetic parameters of biochar degradation and its long-term impact on these enzymatic and thermodynamic processes remain poorly quantified and require further targeted research.

The kinetics of CH_4 and N_2O fluxes can be modeled using Michaelis–Menten kinetics, where the reaction rate is a function of the maximum rate and the half-saturation constant. Biochar amendment effectively increases the maximum rate of methanotrophs by providing a highly porous protective habitat that increases their population density. Simultaneously, WSI ensures that O_2 concentration does not become the rate-limiting substrate. Enzymatically, biochar downregulates the abundance of methyl-coenzyme M reductase (*mcrA*) by altering soil pH and adsorbing labile carbon while upregulating particulate methane monooxygenase (*pmoA*) due to improved aeration [13,19]. For N_2O , biochar promotes the expression of nitrous oxide reductase (*nosZ*), mitigating N_2O spikes during WSI rewetting cycles [38].

Furthermore, the degradation kinetics of biochar itself are biphasic: a rapid initial mineralization of the labile aliphatic fraction, followed by an extremely slow, first-order kinetic decay of the recalcitrant aromatic carbon [42,43]. This slow kinetic decay ensures that the carbon sequestration benefits persist over decades. Future predictive models must incorporate these specific kinetic parameters—such as biochar-induced shifts in for the maximum rate and the half-saturation constant for methanotrophic and denitrifying enzymes—to accurately forecast GHG inventories under combined management [44].

Carbon availability and microbial communities provide a further connection across seasons. Sorption and organo-mineral associations may restrict some substrates, whereas labile carbon introduced with an amendment may support microbial activity and CH_4 production [9,14,37]. Changes in microbial abundance or community composition can be consistent with these explanations but do not alone quantify process rates or establish causation. Interpreting long-term combined effects therefore requires linked measurements

of carbon pools, material properties, redox conditions, microbial processes, and gas fluxes, with treatment history and shared experimental sites accounted for [13,20,40].

5.3. Evidence Synthesis: CH_4 , N_2O , and Net Climate Outcomes Under Combined Management

The mechanistic complementarity between WSI and biochar described above is reflected in field evidence of joint mitigation, although its expression differs among gases and experimental settings. Irrigation alters the duration of reducing conditions, whereas biochar can modify the soil environment in which carbon and nitrogen transformations occur. Their combination may therefore suppress CH_4 while moderating the N_2O response to drying and rewetting, but the available studies also document unchanged or increased emissions [12,18,35–39,41]. A central distinction in interpreting this literature is between the additional contribution of biochar within WSI and the overall performance of a combined treatment relative to conventional flooding. These comparisons address different questions: the former characterizes the amendment response under a given water regime, while the latter evaluates a change in the management system as a whole.

For CH_4 , decreases and increases are best considered as contrasting response groups within a specified comparison. Additional suppression following biochar addition under CI is supported by initial and later observations at Taihu [13,36]. A larger reduction of 75–89% across three years was reported for AWD combined with acid-modified rice-straw biochar but against CF without biochar [41]; it consequently represents the performance of both practices together rather than the amendment alone. In contrast, Pandey et al. [37] reported a 38% increase in CH_4 associated with biochar addition across the studied AWD and permanently flooded regimes. A combined treatment may therefore outperform a flooded or manure-amended baseline while still emitting more CH_4 than the corresponding no-organic-amendment treatment. The apparent contrast between mitigation and increased emissions is meaningful only when these baselines are kept distinct and is consistent with competing effects on aeration, carbon supply, and microbial activity [9,36,37].

For N_2O , decreased emissions are reported across several biochar experiments under WSI, with a descriptive range of 19.3–62% relative to same-irrigation no-biochar controls [12,13,35,38]. However, lower means are not uniformly significant, and nonsignificant biochar responses also occur under AWD, permanent flooding, and a CI dose comparison [13,37]. Moreover, a decrease relative to unamended AWD can coexist with elevated emissions relative to flooding: the AWD-associated increase averaged 210% in one field experiment [35], while emissions remained above CF without biochar despite within-AWD mitigation also being reported [12]. These contrasting directions refer to different management comparisons, not contradictory measurements. Biochar can therefore attenuate the N_2O cost of drying and rewetting without necessarily eliminating that cost or restoring the flooded baseline.

For integrated climate outcomes, the evidence separates into lower GWP and no significant additional improvement. Within-CI reductions in $\text{CH}_4 + \text{N}_2\text{O}$ GWP were reported in second-season and later observations on the Taihu platform [13,39]; the second-season reductions were 35.7% and 21.5% at 20 and 40 t ha⁻¹, respectively [39]. Their agreement provides temporal support at a related site rather than independent geographical replication. By contrast, nonsignificant GWP differences occurred during the first season of that CI experiment [39] and in all three years of a separate within-AWD comparison [12]. The absence of significant additional improvement from combining AWD and biochar in Thailand [18] further cautions against assuming that the benefits of the two practices accumulate. Its separately reported annual GHG reductions of 48.2% for AWD and 8.01% for biochar cannot be added to obtain a combined effect. Thus, reduced emissions in individual gases do not consistently translate into a statistically supported additional climate benefit.

5.4. Environmental and Management Factors Influencing WSI–Biochar Coupling Effects

The variability reflects the interacting influences of hydrology, soil characteristics, biochar properties and dose, nitrogen management, and treatment history. These factors help explain why additional mitigation and maintained crop production occur in some settings but are not consistently reproduced across sites or seasons. The coupling window is therefore best understood as a context-dependent relationship among environmental and management conditions, rather than a fixed set of thresholds or a classification of management outcomes.

Hydrology and soil properties together define the physical setting in which this coupling occurs. Drying interrupts the reducing conditions that sustain CH_4 production, but its duration and timing also influence crop water stress and the transitions associated with N_2O release [4,5,32]. The same irrigation schedule can therefore produce different soil moisture and aeration patterns in soils with contrasting texture and drainage characteristics. Water potential and water-table depth describe related but distinct aspects of this environment, and their implications vary with soil properties, crop stage, and weather. Organic carbon availability and pH further shape the microbial response to these changes [9,32]. Consequently, an irrigation threshold associated with mitigation at one site cannot be interpreted as a general boundary between beneficial and adverse WSI–biochar responses; its significance depends on the soil conditions and nitrogen availability accompanying each drying–rewetting cycle.

The presence or absence of an efficient field drainage system has fundamental bearing on the scope to which WSI and biochar may help to reduce emissions. A good drainage arrangement favors fast water removal when the soil is drying, which promotes aeration and enables oxidation of CH_4 [18]. On the other hand, inadequate drainage tends to maintain long-term waterlogging—even under WSI conditions—thus defeating the purpose of aeration and possibly aggravating CH_4 emissions (if biochar adds labile carbon) [32]. Moreover, a drainage-efficient system plus biochar’s ability to hold water may yield ideal soil moisture during droughts, with less crop stress and fewer N_2O pulses [9,20].

Within this hydrological setting, biochar influences the combined response through a combination of properties rather than application rate alone. Feedstock and production conditions jointly determine alkalinity, sorption behavior, pore structure, and the availability of carbon substrates, with potentially contrasting consequences for CH_4 production and N transformations [9,11,14]. The contrasting CH_4 responses mentioned in Sections 4.1 and 5.3, together with differing dose patterns within CI, show that neither the sign nor the magnitude of the amendment effect can be inferred from application rate alone [13,36,37]. Likewise, results obtained with acid-modified rice-straw biochar [41] describe a distinct material–management combination rather than the response of all crop-residue or woody biochars. Soil pH and carbon status provide an additional context for these material effects: a change that alleviates a limitation in one soil may offer less benefit in another. The relevant principle is therefore matching amendment properties to soil and water conditions, although the available evidence does not yet define a universally effective dose or pyrolysis-temperature class. Hydrochar represents a further material distinction because its production pathway and properties differ from those of pyrolytic biochar [26].

Nitrogen management connects these physical and material effects to the risk of N_2O production. Mineral N availability during aeration and rewetting can alter the balance between N_2O formation and consumption, providing a plausible explanation for variation in the buffering response to biochar [9,20]. Decreased N_2O under AWD has been observed across different mineral-N regimes, including 0 and 180 kg N ha^{−1} [38] and 210 kg N ha^{−1} [35], indicating that mitigation is not confined to a single fertilizer level. These observations do not, however, identify an N threshold above which the amendment ceases to be effective.

Interpretation also depends on the distinction between mineral fertilizer N and total N supplied through biochar or other organic inputs. Similarly, the use of split fertilization in a successful treatment is consistent with coordinated water and nutrient management but does not isolate the contribution of fertilizer timing to a water $\times$ biochar interaction. Thus, N supply is best understood as a moderator of the combined response rather than an independently established cutoff.

Temporal responses likewise encompass additional mitigation and nonsignificant differences, rather than a uniform decline in effectiveness. Lower GWP in second-season and later observations under CI indicates that benefits can emerge or persist beyond the first season [13,39]. Reapplication can also enhance mitigation under particular conditions: annual application of 2.5 t ha⁻¹ for five years reduced cumulative CH₄ by 29.32% relative to a single initial dose of 12.5 t ha⁻¹ under CI [40]. Conversely, nonsignificant responses occurred for first-season GWP after application [39], for the same equal-total-dose frequency comparison under FI [40], and for additional yield-scaled mitigation from reapplication in a rice–wheat rotation [20]. These null findings concern different questions—response onset, application frequency, and yield-scaled performance—and cannot be pooled as one aging effect. In particular, the rotation comparison did not hold cumulative dose equal or include an experimental irrigation factor. Taken together, the evidence suggests that treatment history interacts with water and cropping conditions, not that biochar must invariably be replenished to maintain mitigation.

The interaction among these factors is particularly important when interpreting differences between studies. Additional CH₄ mitigation and season-dependent GWP benefits under CI [13,36,39] coexist in the literature with N₂O buffering but no significant additional GWP benefit under AWD [12], as well as with no significant additional improvement from combining the practices in Thailand [18]. These contrasts are consistent with context dependence, but they do not isolate hydrology, soil properties, or material characteristics as the cause of the differences. Nor does a nonsignificant response establish zero effect or equivalence [8,38].

The broader literature supports this context-dependent interpretation while differing in its capacity to resolve the combined response. Biochar meta-analyses provide a wider perspective on heterogeneity and aging [10,16,17], whereas studies of phosphorus retention, nutrient uptake, water use, and crop performance help explain agronomic responses [10,16,17]. Such evidence complements the direct emission evidence synthesized in Section 5.3, but improvements in nutrient retention or yield alone do not establish a reduction in CH₄ + N₂O GWP. Similarly, regional scaling and nitrogen-management scenarios address the potential reach of mitigation [45,46] rather than directly demonstrating a water $\times$ biochar interaction.

Overall, the effectiveness of WSI–biochar management depends on how water conditions, soil properties, amendment characteristics, N availability, and treatment history operate together. The evidence supports conditional benefits while retaining increases and nonsignificant responses and does not yet define a common set of quantitative boundaries that predicts outcomes across sites. Long-term factorial studies across contrasting soils are needed to separate these influences and assess whether locally observed benefits persist under changing environmental and management conditions.

6. Limitations and Uncertainty Analysis

6.1. Representativeness and Transferability Across Time and Space

The evidence synthesized in Sections 3–5 supports conditional mitigation benefits, but its representativeness differs between the individual practices and their combination. Broad syntheses of WSI or biochar provide useful context for response variability [4,9,11,32], yet their main-effect estimates cannot establish a water $\times$ biochar interaction. Selective

publication or incomplete reporting of unfavorable outcomes may also affect the evidence base; their extent has not been quantified in this review. Retaining emission increases and nonsignificant additional benefits is therefore essential to avoid presenting successful treatments as representative of all settings.

Temporal uncertainty concerns the persistence and timing of responses rather than an assumed loss of effectiveness. First-year syntheses [14], later-season observations under CI [13,39], rotation-based reapplication evidence [20], and a five-year application-frequency experiment [40] address different aspects of treatment history. Together, they indicate that later benefits, delayed responses, and nonsignificant differences can coexist without demonstrating that aging necessarily diminishes or reverses mitigation. Separating aging from cumulative dose, reapplication, cropping history, and interannual weather remains difficult. Spatial transferability likewise requires validation: an irrigation threshold established under one combination of soil, weather, and nitrogen management cannot be assumed to provide equivalent moisture conditions, crop protection, or emission outcomes elsewhere [4,5,32].

The geographic coverage and comparability of the combined-management evidence remain insufficient for regional effect-size estimates or risk rankings. Studies differ in irrigation controls, biochar materials, nitrogen inputs, and observation periods, while repeated observations on the Taihu platform provide temporal evidence rather than independent geographic replication [13,36,39]. Consequently, between-study contrasts cannot isolate climate or geography as the cause of a response, nor can they establish that an entire climatic zone has higher CH₄ mitigation potential or lower N₂O and yield risks. The treatment-specific findings synthesized in Section 5.3, including increases and nonsignificant outcomes [12,18,37], remain relevant to local assessment, but their transfer requires independently replicated, comparable trials across contrasting environments.

6.2. Measurement and Accounting-Boundary

Measurement uncertainty is particularly important when water and nitrogen management generate short-lived emission events. Sampling that misses drainage, fertilization, or rewetting events can fail to capture N₂O peaks, but the direction and size of error in seasonal estimates depend on event timing, sampling coverage, and flux interpolation. Routine observations should therefore be complemented by event-focused measurements, with sampling frequency and calculation procedures reported explicitly. Joint CH₄ and N₂O measurement is also important because the WSI literature includes both increased N₂O and concurrent decreases in the two gases [5,8].

Even with well-performed field measurements, generalizations tend to be boundary-dependent. Net GWP and GHGI outcomes may vary greatly depending on coefficient choice and system boundaries, especially when transitioning from field-level analysis to full life-cycle assessments involving upstream emissions of biochar pyrolysis, of irrigation energy, and of fertilizer production [12,45,47]. Climate accounting introduces further uncertainty when studies use different gases, conversion factors, observation periods, or system boundaries. Field CH₄ + N₂O GWP, CH₄-only GHGI, soil-carbon-adjusted net balances, and life-cycle assessments answer different questions and should not be treated as interchangeable indicators [12,41].

6.3. Farm-Scale Feasibility Co-Risks, Scaling-Model Uncertainty, and Harmonized Reporting

The experimental feasibility of a treatment does not establish its affordability or operational suitability. The doses span different materials and management histories, but their practical implications depend on delivered biochar cost, transport, incorporation, labor, and access to reliable water control. Financial analysis of rice-production scenarios illustrates the importance of evaluating economic and emission outcomes together [47],

rather than providing a universal cost threshold for WSI–biochar management. Farm-scale feasibility therefore remains a site-specific question, particularly where smallholders cannot independently control drainage or bear the upfront cost of amendment.

Environmental safeguards likewise require attention beyond CH₄, N₂O, and yield. Evidence concerning grain metal uptake under drainage [7] and phosphorus retention and leaching with acid-modified biochar [16] shows why the outcome set should extend beyond climate indicators; it does not demonstrate that every combined treatment creates the same co-risks. Material quality, soil conditions, and nutrient management should guide monitoring of contaminants, NH₃ volatilization, nitrate or phosphorus losses, and relevant agrochemical interactions. These endpoints should be assessed as potential benefits or adverse effects, not presumed consequences of biochar addition.

Scaling adds uncertainty when local responses are extended beyond their calibration domain. Regional WSI assessments [45] and combined-management modeling [19] can organize scenarios and identify data needs, but they do not substitute for independent field validation of the combined response. Model structure, parameter uncertainty, and assumptions about adoption should remain distinguishable from measured treatment variability. The priorities summarized in Table 2 consequently link each uncertainty to the evidence needed to reduce it, providing a basis for locally evaluated management rather than a claim that standardized reporting alone makes deployment reliable.

Table 2. Key uncertainties and evidence needed to strengthen WSI–biochar assessment.

Uncertainty Domain *	Main Limitation	Priority Response
Temporal persistence [13,14,20,39,40]	Response onset, aging, cumulative dose, and reapplication are difficult to separate	Maintain untreated controls and multi-season observations; distinguish equal-total-dose frequency tests from additional-dose treatments
Representativeness and independence [4,9,11,13,36,39]	Heterogeneous sites and repeated observations from shared platforms limit transferability	Document experimental provenance and obtain independent replication across contrasting environments
Measurement [5,8,24]	Sparse or poorly timed sampling can miss transient emissions	Combine routine and event-focused measurements; report coverage, interpolation, and missing-data handling for both gases
Climate accounting [12,41]	Gases, GWP factors, yield denominators, and carbon-accounting boundaries differ	Declare the metric and boundary; separate field emissions, carbon storage, and life-cycle components and test key assumptions
Economic and operational feasibility [47]	Experimental performance may not translate into affordable, controllable farm practice	Assess local costs, labor, water-control access, and risk-adjusted returns alongside climate outcomes
Environmental co-risk [7,16]	Climate indicators do not capture all nutrient, material-quality, or food-safety outcomes	Select locally relevant co-risk indicators and evaluate both benefits and adverse responses
Scaling/model [19,45]	Under-calibrated scenario models	Use independent validation and separate model, parameter, and adoption uncertainty

* The practices in Table 2 are proposed responses to evidence limitations, not validated thresholds or guarantees of uncertainty removal; references identify relevant evidence contexts rather than proof of every proposed practice.

7. Future Research Priorities

7.1. Long-Term Evidence Networks and Quantitative Coupling Boundaries

Future progress should shift from broad efficacy demonstrations toward research capable of resolving persistence, transferability, and conditions governing combined responses. A primary priority is coordinated long-term monitoring across contrasting rice-growing environments, using harmonized treatment definitions and routine sampling complemented by observations during drainage, fertilization, and rewetting. These networks should jointly track CH₄, N₂O, yield, SOC, and biochar-aging trajectories under repeated wet–dry cycles. The contrasting temporal questions addressed by later-season observations, reapplication studies, and the five-year application-frequency experiment [13,20,39,40] show the value of monitoring beyond initial seasons, preferably for five years or longer where feasible. Such monitoring would help distinguish persistent, delayed, and declining responses without assuming that aging necessarily weakens mitigation. Independent sites, repeated seasons, cumulative dose, and application history should be recorded separately so that apparent persistence is not confused with geographic replication or additional amendment input.

In parallel, factorial experiments should translate the context-dependent relationships discussed in Section 5.4 into quantitative, testable response boundaries. Priority combinations include AWD intensity × biochar dose, AWD intensity × nitrogen timing or form, and biochar property profiles × soil classes. Nested water, biochar, and nitrogen treatments could help distinguish an amendment effect from changes in nutrient supply, while replicated experiments across soil classes would test whether material properties explain differences in response. Mapping CH₄, N₂O, yield, and water-use responses together would identify candidate conditions for additional mitigation and agronomic trade-offs, rather than presupposing an optimal coupling window. Explicit controls and a stated interaction scale are essential for separating package effects, within-WSI amendment effects, and statistical interactions. Long-term replication and independent tests of candidate boundaries can then move the field from descriptive heterogeneity toward predictive relationships with a defined domain of applicability.

7.2. Mechanistic Attribution, Cross-Site Calibration, and Decision-Support Tools

Mechanistic attribution should be strengthened by pairing flux measurements with diagnostics of redox dynamics, microbial functional indicators, and carbon–nitrogen turnover. Measurements of available carbon and mineral nitrogen alongside biochar properties would help test how aging, substrate supply, and wet–dry transitions influence CH₄ production and N₂O formation or consumption. Event-focused observations should be linked to seasonal gas balances to establish whether transient process changes explain cumulative outcomes. These relationships may also provide candidate indicators of N₂O pulses or diminished CH₄ abatement, but their predictive value requires testing beyond the observations used to derive them; microbial associations alone do not establish causal pathways or thresholds. Agronomic and nutrient-retention findings [10,16,17] can inform these hypotheses while remaining distinct from evidence of a complete climate benefit.

Building on this foundation, cross-site datasets could support parameter libraries stratified by measured soil, climate, and management characteristics. Candidate entries include irrigation-response ranges, biochar property–soil relationships, nitrogen–water timing interactions, and responses to unusual rainfall or interrupted water delivery. The aim would be to test which relationships transfer between environments, not to assign prescriptions solely from a climatic-zone label. Regional WSI assessments [45] and combined-management modeling [19] provide starting points for linking field observations with process-based predictions, with calibration and independent validation kept separate. Shared experimental platforms and overlapping publications should remain identifiable, and synthesis should

harmonize comparators and accounting boundaries before pooling responses. Where comparable data permit, estimates of heterogeneity and prediction uncertainty would be more informative for transfer than observed extrema alone.

To connect field science with practical evaluation, this evidence base could inform low-cost digital decision-support tools, including event-triggered pulse-risk alerts and integrated emission–yield–cost dashboards. Research should assess predictive accuracy, usability, data requirements, and cost under realistic farm conditions. Irrigation records, water-status sensors, and remote observations may supply useful inputs, but neither water-status sensing nor canopy imagery directly verifies N₂O pulses. Predicted risks therefore need to remain distinguishable from measured emissions, and any use in monitoring, reporting and verification should be evaluated against independent gas observations or validated estimation methods. The research objective is to develop transparent, locally interpretable tools that disclose uncertainty and flag conditions outside their tested domain.

7.3. Economic Feasibility, Equity, and Risk-Inclusive System Assessment

Affordability, operational acceptability, and the distribution of benefits and burdens should be integrated into experimental and modeling designs from the outset. Financial evaluation of rice-production scenarios illustrates the relevance of examining economic and emission outcomes together [47]. Future studies should assess marginal abatement cost, risk-adjusted returns, labor and infrastructure constraints, delivered amendment costs, and alternative uses of residues. Comparisons between one-time and phased dosing should state whether cumulative input is equal, distinguishing application frequency from additional material. Lower-input strategies such as localized root-zone application, biochar-based fertilizers, and seed coating offer specific research directions for addressing the cost of bulk amendment. Whether they reduce material requirements while retaining climate and agronomic benefits remains a question for direct comparison with conventional incorporation and appropriate no-biochar controls, using joint CH₄ and N₂O measurements, yield, and cost assessment.

Equity analysis should be treated as a core component rather than an add-on, clarifying who captures benefits and who bears transition costs under different farm structures and policy instruments. Participatory studies could examine how farm size, tenure, irrigation access, and financial support affect adoption, labor burdens, and exposure to production risks. System assessment should also track locally relevant co-risks, including material contaminants, NH₃ volatilization, nitrate and phosphorus losses, grain safety, and agrochemical interactions. Evidence on grain metal uptake and phosphorus retention [7,16] motivates this broader outcome set without implying that every combined treatment produces the same benefits or adverse effects. Linking field data, process-based models, and life-cycle methods would help bridge plot-level findings to farm, basin, and policy scenarios under realistic adoption assumptions, with explicit treatment of energy use, transport, carbon persistence, and potential double counting.

Three priorities are summarized together in Table 3. Harmonized observation provides a common foundation, long-term and mechanistic studies test persistence and transferability, and economic and risk-inclusive assessment establishes practical relevance. These activities should inform one another rather than follow a fixed timetable leading inevitably to deployment. Their contribution is to establish when combined management offers a credible additional benefit, when further evidence is needed, and when amendment may not be justified.

Table 3. Integrated research priorities for combined WSI and biochar management.

Research Priority	Key Questions	Priority Approaches	Intended Contribution
Long-term networks and response boundaries	How do combined responses persist, change, and vary with water, material, nitrogen, and soil conditions?	Independent multi-site observations of CH ₄ , N ₂ O, yield, SOC, and aging; routine and event-focused sampling; factorial response mapping with explicit controls and application histories	Temporal trajectories and candidate response boundaries, with interaction estimates and uncertainty tested beyond the original sites
Mechanistic attribution and decision support	Which processes explain response variability, and which relationships can predict outcomes elsewhere?	Linked flux, redox, and carbon–nitrogen diagnostics; cross-site parameter libraries; independent model validation; evaluation of pulse-risk alerts and emission–yield–cost tools	Tested process explanations and locally interpretable predictions with stated applicability limits, distinct from direct emission verification
Economics, equity, and system assessment	Under what resource, adoption and environmental conditions is combined management worthwhile?	Marginal abatement cost and risk-adjusted returns; equal-input frequency comparisons and lower-input strategy trials; participatory equity assessment; co-risk monitoring and explicit life-cycle accounting	Evidence on affordability, benefit distribution, and net climate outcomes for adoption, modification, or non-adoption under realistic constraints

8. Conclusions

Water-saving irrigation and biochar offer complementary opportunities to reduce greenhouse gas emissions from paddy rice, but their combination is not inherently superior to either practice alone. WSI generally suppresses CH₄ by interrupting prolonged anaerobic conditions, while its N₂O response depends on the water and nitrogen regime. Biochar can moderate this trade-off through changes in the soil environment and carbon–nitrogen transformations, yet its additional mitigation benefit is inconsistent. The central conclusion is therefore that combined management has conditional potential, rather than a universally synergistic effect. Evidence indicates that combining AWD with biochar application rates of 10–40 t ha^{−1} can reduce CH₄ emissions by 43–52% and N₂O emissions by 19–62% compared to continuously flooded controls without biochar, ultimately lowering the global warming potential by 37–47%.

Whether this potential is realized depends on the interaction of water regime, soil characteristics, biochar properties and dose, nitrogen management, and treatment history. These interacting factors favor locally evaluated combinations over uniform prescriptions. Their value should be judged by the joint response of CH₄ and N₂O alongside yield and water use, with a clear distinction between adding biochar to an existing WSI regime and replacing a flooded system with combined management. Favorable field emissions alone are insufficient to establish a net climate benefit or practical suitability; carbon persistence, production and transport impacts, costs, and environmental co-risks also matter.

The principal research need is to determine where combined benefits arise, how long they persist, and whether they remain worthwhile beyond experimental plots. Independent, long-term, multi-site factorial studies linking emission measurements with mechanistic, agronomic, and system-level assessment are essential to resolve these questions. Progress toward climate-smart rice production will depend on identifying combinations that deliver sustained climate benefits while maintaining crop production and remaining feasible under

local conditions. Specifically, future studies must quantify biochar degradation kinetics and determine thermodynamic parameters to better predict long-term efficacy.

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