Greenhouse Gas Emissions from Salt-Affected Soils: Mechanistic Understanding of Interplay Factors and Reclamation Approaches

Ram K. Fagodiya 1,*, Sandeep K. Malyan 2, Devendra Singh 3, Amit Kumar 4,5, Rajender K. Yadav 1, Parbodh C. Sharma 1,6 and Himanshu Pathak 6

1 ICAR-Central Soil Salinity Research Institute, Karnal 132001, Haryana, India
2 Dayal Singh Evening College, University of Delhi, Lodi Road, New Delhi 110003, Delhi, India
3 ICAR-Central Arid Zone Research Institute, Jodhpur 342003, Rajasthan, India
4 Central Muga Eri Research and Training Institute, Central Silk Board, Jorhat 785700, Assam, India
5 Central Sericultural Research and Training Institute, Central Silk Board, Mysore 570008, Karnataka, India
6 Indian Council of Agricultural Research, New Delhi 110001, Delhi, India
* Correspondence: ram.iari4874@gmail.com (R.K.F); pcsharma.knl@gmail.com (P.C.S.)

Abstract: Salt-affected soils contain high levels of soluble salts (saline soil) and exchangeable sodium (alkali soil). Globally, about 932 million ha (Mha), including 831 Mha of agricultural land, is salt-affected. Salinity and sodicity adversely affect soil microbial diversity and enzymatic activities, and thereby carbon and nitrogen dynamics and greenhouse gas (GHG) emissions from soils. In this review article, we synthesize published information to understand the impact of salinity and sodicity on GHG production and emissions from salt-affected soils, and how various reclamation amendments (gypsum, phosphogypsum, organic manure, biochar, etc.) affect GHG emissions from reclaimed soils. Nitrous oxide (N\textsubscript{2}O) and methane (CH\textsubscript{4}) emissions are of greater concern due to their 298 and 28 times higher global warming potential, respectively, compared to carbon dioxide (CO\textsubscript{2}), on a 100-year time scale. Therefore, CO\textsubscript{2} emissions are given negligible/smaller significance compared to the other two. Generally, nitrous oxide (N\textsubscript{2}O) emissions are higher at lower salinity and reduced at higher salinity mainly due to: (a) higher ammonification and lower nitrification resulting in a reduced substrate for denitrification; (b) reduced diversity of denitrifying bacteria lowered down microbial-mediated denitrification process; and (c) dissimilatory nitrate reduction to ammonium (DNRA), and denitrification processes compete with each other for common substrate/nitrate. Overall, methane (CH\textsubscript{4}) emissions from normal soils are higher than those of salt-affected soils. High salinity suppresses the activity of both methanogens (CH\textsubscript{4} production) and methanotrophs (CH\textsubscript{4} consumption). However, it imposes more inhibitory effects on methanogens than methanotrophs, resulting in lower CH\textsubscript{4} production and subsequent emissions from these soils. Therefore, reclamation of these soils may enhance N\textsubscript{2}O and CH\textsubscript{4} emissions. However, gypsum is the best reclamation agent, which significantly mitigates CH\textsubscript{4} emissions from paddy cultivation in both sodic and non-sodic soils, and mitigation is higher at the higher rate of its application. Gypsum amendment increases sulfate ion concentrations and reduces CH\textsubscript{4} emissions mainly due to the inhibition of the methanogenesis by the sulfate reductase bacteria and the enhancement of soil redox potential. Biochar is also good among the organic amendments mitigating both CH\textsubscript{4} and N\textsubscript{2}O emissions from salt-affected soils. The application of fresh organic matter and FYM enhance GHG emissions for these soils. This review suggests the need for systematic investigations for studying the impacts of various amendments and reclamation technologies on GHG emissions in order to develop low carbon emission technologies for salt-affected soil reclamation that can enhance the carbon sequestration potential of these soils.

Keywords: salt-affected soils; GHG emissions; soil salinity; soil sodicity; gypsum; phosphogypsum
1. Introduction

Soil salinization and sodification are serious causes of land degradation particularly in arid and semi-arid regions worldwide. Soils with high levels of soluble salts (saline soil) and exchangeable sodium (alkali soil) are considered salt-affected soils [1]. Globally, 831 million ha (Mha) of the land of which ~20% is agricultural and ~33% is irrigated, is distributed among 120 countries and is salt-affected [2,3]. The expansion of these soils is expected to increase due to climate change, intrusion of sea water in coastal regions, and poor irrigation management in canal command areas [4]. Enhanced intensity and frequency of extreme events, particularly storms/cyclones in coastal areas, have been observed during the last 50 years [5]. Additionally, unjustifiable use of groundwater, excess use of synthetic fertilizers, and poor soil management are the causes of salt-induced soil degradation [6]. The saline (with excess salt) and alkali (with high residual alkalinity) groundwater are generally associated with the development of salt-affected lands [7]. Globally, 2400 Mha of land area (16% of total land) is underlain with the saline/alkali groundwater at the shallow/intermediate depth and the maximum area (14% of total saline/alkali water area) is found in the Basin of West and Central Asia [7]. The changing environmental scenario would further reduce the availability of good quality waters for irrigations [5], which will further enhance the utilization of marginal quality waters for irrigation and consequently enhance the expansion of area under salt-affected soil [8].

The enhanced greenhouse gas (GHG) emissions are the real cause of the greenhouse effect and agriculture contributes significantly [9] towards emissions through key processes/managements such as methane (CH$_4$) from enteric fermentation and rice cultivation, nitrous oxide (N$_2$O) from the application of synthetic fertilizers, and carbon dioxide (CO$_2$) from tillage operations [10,11]. The emissions of N$_2$O and CH$_4$ are of greater concern due to 298 and 28 times higher global warming potential (GWP) than that of CO$_2$ [10] on a 100-year time scale, respectively [5]. Global GHG emissions from agricultural activities were 9.3 Giga tonnes (Gt) of CO$_2$ equivalents (CO$_2$ eq) in 2018 [2]. The contribution of CH$_4$ and N$_2$O emissions from crops and livestock was 5.3 Gt CO$_2$ eq with agricultural soils and enteric fermentation being major sources contributing 39.5 and 39.2%, respectively [2]. The emissions of CO$_2$, CH$_4$, and N$_2$O occur from the agricultural soils through microbial-mediated processes/pathways. CO$_2$ flux from agricultural soils can be due to (i) soil respiration (root and microbial respiration), (ii) ecosystem respiration, and (iii) net ecosystem exchange (NEE), i.e., the difference between plant photosynthesis and ecosystem respiration (heterotrophic, as well as autotrophic) [12]. Under anoxic conditions, CH$_4$ is produced by methanogens and consumed by methanotrophs under oxic and anoxic conditions [10,13–15]. N$_2$O is produced mainly through the denitrification process in anaerobic environments and the nitrification (hydroxylamine oxidation and nitrifier denitrification) process in aerobic environments [16–18].

Irrigation with saline/sodic waters induces changes in soil structure and adversely affects the microbe-mediated soil processes [19]. Generally, the excess salt in soils restricts the microbial population and their activity through osmotic stress [20]. High salt concentration in soil inhibits the soil organic matter decomposition through alteration of microbial activities leading to either decrease or increase in mineralization of carbon (C) and nitrogen (N) [21,22]. However, inhibition of N mineralization is temporary and recovers at later stages [23]. GHG emissions from soils are governed by microbial activities involved in organic matter decomposition, nitrification, denitrification, methanogenesis, and CH$_4$ oxidation processes, and salinity and sodicity have significant effects on these processes [24]. Usually, GHG emissions decrease with increased soil salinity and sodicity. A decrease in N$_2$O [25,26], CH$_4$ [27,28], and CO$_2$ [29,30] emissions with increased salinity and sodicity has been reported in several studies. However, reports on increased N$_2$O [30], CH$_4$ [24,31], and CO$_2$ [31] emissions are also available. To our best knowledge, the review article concerning the N$_2$O, CH$_4$, and CO$_2$ emissions from the salt-affected soils, various factors affecting their emissions, and the impacts of reclamation approaches of salt-affected soils on GHG emission has not yet been published. Therefore, the present manuscript has
been organized to understand the conditions, interplaying factors along with the impact of reclamation processes on the GHG emissions from the salt-affected soils to refine the reclamation practices and ecosystem sustainability along with the climate change mitigation in these affected agro-ecosystems.

2. Salt-Affected Soils, Global Extent, and Distribution

Salt-affected soils have a high concentration of soluble salts in such a quantity that negatively affect normal growth and productivity [32]. These problematic salts are mainly carbonates (CO$_3^{2-}$), bicarbonates (HCO$_3^-$), chlorides (Cl$^-$), and sulfates (SO$_4^{2-}$) of sodium (Na$^+$), calcium (Ca$^{2+}$), and magnesium (Mg$^{2+}$) [33]. Salt-affected soils are classified into three categories, i.e., (i) saline, (ii) alkali/sodic, and (iii) saline-alkali/saline-sodic soils based on the electrical conductivity of saturated paste extract (EC$_e$), pH of saturated paste (pH$_s$), exchangeable sodium percentage (ESP), and sodium adsorption ratio in saturated paste extract (SAR$_e$) [33]. Saline soils contain high concentration of neutral salts mainly chlorides and sulfate of sodium, calcium and magnesium, and have EC$_e$ > 4 dS m$^{-1}$, pH$_s$ < 8.5, ESP < 15, and SAR$_e$ < 13 [34]. Alkali/sodic soils possess excess contents of carbonates, bicarbonate and silicate salts of sodium, and characterized by EC$_e$ < 4 dS m$^{-1}$, pH$_s$ > 8.5, ESP > 15, and SAR$_e$ > 13. While saline-alkali soils have high levels of both soluble salts and ESP, and these show EC$_e$ > 4 dS m$^{-1}$, SAR$_e$ > 13, ESP > 15, and variable pH$_s$ due to collective effects of both salinity and sodicity [35]. Detailed characteristics of salt-affected soils are given in Table 1. Globally, about 20% of agricultural land (831 Mha) is salt-affected [3]. Out of the total salt-affected soils, 47.8% (397 Mha) is saline and 52.2% (434 million ha) is sodic in nature (Figure 1). The area of these salt-affected soils is distributed in 120 countries and represents 10, 20, and 33% of the global, agricultural, and irrigated lands, respectively [33,36]. Globally, the Amu-Darya and Syr-Darya River Basins (Aral Sea Basin) in Central Asia, the Indo-Gangetic Basin in India, the Indus Basin in Pakistan, the Yellow River Basin in China, the Euphrates Basin in Syria and Iraq, the Murray-Darling Basin in Australia, and the San Joaquin Valley in the United States are the well-known regions where salinization is extensively reported [37]. The highest (~340 Mha) salt-affected area, i.e., 50% of total global sodic soils, is found in Australia, followed by Central and South Asia (~212 Mha) [38,39]. Kazakhstan (~60 Mha) and Uzbekistan (~28 Mha) are the main salinity-affecting countries in the Central Asian region [39,40]. In India, the problem extends over an area of about 6.73 Mha (2.96 Mha saline soil, and 3.77 Mha sodic soil) land, which is about 2% of India’s total geographic area (TGA) [41,42].

![Figure 1. Global area of salt-affected soils and their distribution in the different regions of the world](Source: FAO and ITPS [43]).
Table 1. Types/classes of salt-affected soils and their chemical characteristics.

| Salinity Class            | EC<sub>e</sub> (dS m<sup>-1</sup>) | ESP | (pH<sub>s</sub>) | (SAR<sub>e</sub>) | Type of Dominant Salts                                                                 | Problems Associated                                                                 |
|---------------------------|----------------------------------|-----|----------------|-----------------|--------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------|
| Saline soils              | >4                               | <15 | <8.5           | <13             | High levels of soluble salts of chlorides (Cl<sup>-</sup>) and sulfate (SO<sub>4</sub>²<sup>-</sup>) of sodium (Na<sup>+</sup>), calcium (Ca<sup>2+</sup>), and magnesium (Mg<sup>2+</sup>) | Hinder water absorption by plants due to high osmotic effects. Possible toxicity and antagonism of dominant ions at higher electrical conductivity |
| Alkali/sodic Soils        | <4                               | >15 | >8.5           | >13             | High concentration of carbonate (CO<sub>3</sub>²<sup>-</sup>) and bicarbonate (HCO<sub>3</sub>⁻) salts of sodium (Na<sup>+</sup>) in soil solution, and Na<sup>+</sup> on exchange sites | Sodium, Carbonate, and bicarbonate toxicity to plants. Dispersion of soil structure due to high sodium content. Slaking, swelling, and hard setting of soil surface. Seasonal waterlogging |
| Saline-alkali/ sodic soils| >4                               | >15 | Variable       | >13             | The combined effect of excess salts and high exchangeable sodium percentage           | Hinder water and nutrients uptake due to high osmotic effects. Sodium, carbonate, and bicarbonate toxicity to plants. Dispersion of soil structure due to high sodium content. Slaking, swelling, and hard setting of soil surface |

EC<sub>e</sub> = electrical conductivity of saturated paste extract; pH<sub>s</sub> = pH of saturated paste; ESP = exchangeable sodium percentage; SAR<sub>e</sub> = sodium adsorption ratio in saturated paste extract.

3. Microbial Response to Salinity and Sodicity

Soil microbial communities perform an essential role in the organic matter decomposition, nutrient cycling, and GHG production/consumption and both Soil salinity/sodicity adversely affect the microbial biomass [44] and play a decisive role in the structuring, and distribution of microbial communities [45]. Siddikee et al. [46] reported the adverse impact of soil salinity/sodicity on microbial activities and biogeochemical processes that are essential in the mineralization of nutrients. The salinity/sodicity has an extensive impact on GHG production and emissions from agricultural soils mainly due to their influences on the growth and activities of nitrifying, denitrifying, methanogen and methanotrophs. The mechanisms which may interpret the relationship between salinity/sodicity and these microorganisms are described and depicted in Figure 2. A high concentration of soluble salts inhibits microbial growth due to the adverse impact of increased osmotic stress in general and specific ion toxicity in particular [19]. Soil salinity reduces microbial activities by altering the soil’s physicochemical properties [47]. Increased sodicity decreases the O₂ diffusion in the soil due to the blocking of soil pores and consequently decreases soil respiration [48]. At a high level of soil salinity, the toxicity of Cl<sup>-</sup> and hydrosulfide (HS<sup>-</sup>) ions caused adverse impacts on microbial growth and thereby on N₂O and CH₄ emissions [49].

High salinity levels inhibit the nitrification rate and decrease the availability of nitrate (NO₃<sup>-</sup>) which further limit the denitrification process and thereby reduces the N₂O emissions [50]. Rysgaard et al. [51] reported that an increase in salinity decreases the ammonia adsorption by soil sediments which potentially enhances the availability of ammonia and thus stimulates nitrification [51]. Higher salinity limits the availability of soil organic matter/organic substrates for heterotrophic bacteria and alters the abundance and activities of microbes [52]. It also limits the biodegradation of complex organic substrates into simpler ones (H₂, formate, acetate, alcohol, and other compounds) which are used by methanogens for CH₄ production and thereby reduces CH₄ emissions [53]. The decrease in microbial enzymes and gene activities involved in organic matter decomposition, nitrification, denitrification, methanogenesis, and CH₄ oxidation has also been reported [54]. Increasing salinity enhance the ability of dissimilatory nitrate reduction to ammonium (DNRA) to overcome the denitrification [55]. Because increasing salinity enhances the population and
activity of sulfate reducers [56] and consequently may increase HS$^-$ ions. These HS$^-$ ions have a more inhibitory effect on denitrifying bacteria than DNRA [57].

Figure 2. Response of nitrifying, denitrifying, methanogens, and methylotrophs/methanotrophs involved in GHG production and consumption to soil salinity and sodicity. Orange (nitrification, denitrification, methanogenesis, and methylothrophy), blue (nitrification/denitrification), and green (methanogenesis/methylotrophy) colors represent the GHG production process/pathways affected by salinity and sodicity.

3.1. Effect of Salinity/Sodicity on Nitrification/Denitrification

3.1.1. Nitrification

Nitrification is a two-step process involving (a) ammonia oxidation regulated by ammonia-oxidizing archaea (AOA) and (b) nitrite oxidation process regulated by ammonia-oxidizing bacteria (AOB) [18]. The response of AOA and AOB to salinity level is controversial. Some studies advocated that nitrification is predominated by AOA at salinity levels up to 10–20 parts per thousand (ppt) while activities of AOB decreased at increased salinity [58]. Other investigations advocate that the activities of AOB were more than AOA at increased salinity levels [59]. These contradictory observations indicate that besides the salinity/sodicity levels other factors also affect the community structure and predominance of AOA and AOB. Slightly acidic to neutral pH and availability of ammonia favor the growth of AOB as compared to AOA [60,61]. Guo et al. [62] reported that saline water irrigation increased soil salinity and ammonical N and lower AOA/AOB ratios. Kaushik and
Sethi [63] observed a significant reduction in the growth of both ammonium oxidizers and nitrite oxidizers in rice rhizosphere under increased salinity, and ammonium oxidizers were found more susceptible than nitrite oxidizers to salt stress. Magalhães et al. [64] observed a higher nitrification rate with increasing salinity from 0 to 15 ppt in the Douro River estuary. Zhou et al., [36] found that the growth and activities of nitrifiers were optimum at 5–10 ppt and excess levels of salinity (>10 ppt) showed the inhibitory effect. Overall, moderate salinity enhanced the nitrification rate while excess salinity decreases. Soil moisture is a key factor influencing nitrification and denitrification rates at higher salinity levels [65]. Denitrification had a significantly positive relationship with soil moisture and it increased with an increase in soil moisture when the soil water content was less than 27.03% and decreased with an increase in soil moisture when the soil water content was more than 27.03% [66].

3.1.2. Denitrification

Denitrification is a microbial-mediated multi-step anaerobic process in which nitrate (NO$_3^-$) is sequentially reduced into nitrite (NO$_2^-$), nitric oxide (NO), nitrous oxide (N$_2$O), and finally into atmospheric nitrogen (N$_2$) [67]. Denitrification is facilitated by a combination of four independent enzymes, i.e., nitrate reductase (NAR/NAP), nitrite reductase (NIR), NO reductase (NOR), and N$_2$O reductase, which have been encoded by narGH/narA, nirK/nirS, nonB, and nosZ genes, respectively [68]. Denitrifying bacteria perform differentially under saline and sodic environments. Several studies advocate that soil salinization consistently reduces the denitrification [36,69]. However, Franklin et al. [52] observed an abundance of denitrifiers at low salinity (5 ppt). The negative association of denitrifiers with sediment salinity (0–36 ppt) in estuaries has been explored by Giblin et al. [50] and Santoro et al. [49]. Zhang et al. [70] envisage that increased salinity increases nitrification and denitrification below the threshold value of salinity (EC = 1.13 dS m$^{-1}$) and decreases nitrification and denitrification above the threshold salinity.

Santoro et al. [49] found that the diversity of two genes nirS and nirK was negatively associated with salinity in coastal aquifers and the relative abundance of nirS was higher than nirK. This indicated the environmental effect of salinity on the metabolic performance of the microbial populations. Wang et al. [69] advocated that salinity significantly reduced the abundance of nirK, nirS, and nosZ genes and decreased the population of denitrifying bacteria. Shao et al. [71] found increased N$_2$O efflux under >0.5% salinity conditions. Fiedler et al. [72] reported a 42 times higher abundance of nitrite reductase nirS in salt–affected soil than in controlled soil which suggests that the saline soils have a higher potential for denitrification. N$_2$O reductase (nosZ) is more sensitive to soil salinity and depressed significantly under salinity conditions hence N$_2$O is not reduced into N$_2$ resulting in more N$_2$O accumulation and effluaxes from the denitrification [73]. Decoupling of nitrification or denitrification processes under saline conditions might be a possible mechanism for increasing N$_2$O effluaxes under salinity [71]. However, some other investigations also concluded that N$_2$O emissions are negatively associated with soil salinity [74].

3.2. Effect of Salinity/Sodicity on Methanogenesis/Methanotrophs

Methanogenesis is the microbial-mediated process of CH$_4$ production from the complex organic material by bacteria and archaea. Bacteria hydrolyze the complex organic material into simpler substrates (H$_2$, formate, certain alcohols, acetate, etc.) which are further consumed by methanogens as food and produced CH$_4$ [75]. Limited literature is available about the abundance and community structure of methanogens and methylo trophs in the hypersaline environment. Ollivier et al. [76] reported generation of CH$_4$ from H$_2$ + CO$_2$ at salinities up to 240 ppt but not from acetate even at salinities > 60 ppt. Scholten et al. [77] investigated the abundance and distribution of methanogen-specific functional genes (mcrA) in hypersaline environments. Investigations by Bebout et al. [78] and Smith et al. [79] did not report any significant contribution of methanogenesis in carbon remineralization under a hypersaline environment. This might be due to the predomi-
nance of sulfate reducers or high oxygen concentration in a photic zone that precludes methanogenic activity from such environments under field conditions [78,79].

Heyer et al. [80] reported that the culture of methanotrophic bacteria is capable of growing to a 15% NaCl level. Nguyen et al. [81] reported very low methane emissions from paddy cultivated in salt-affected but cow manure addition enhanced CH$_4$ emissions significantly due to improved relative abundance of methanogens by enhancing soil properties and nutrient availability. Shao et al. [71] and Xiao et al. [82] concluded that salinity imposes more inhibitory effects on methanogens than methanotrophs or methylotrophs. Low salinity favors the growth of methanogens because Na$^+$ ions are required by methanogens for growth, amino acid transportation, methanogenesis, and internal pH regulation [83]. Weston et al. [84] investigated that acetoclastic methanogens were significantly inhibited by the in-situ addition of saline solution in the Delaware Wetland of New Jersey. However, hydrogenotrophic methanogens did not show any significant change in methanogenesis rate under the same investigation.

4. GHG Emissions from Salt-Affected Soils

4.1. Nitrous Oxide Emissions

N$_2$O is produced through nitrification (conversion of NH$_4^+$ to NO$_2^-$, and NO$_2^-$ to NO$_3^-$) and denitrification (conversion NO$_3^-$ to N$_2$) pathways (Figure 3a) [67]. The N$_2$O efflux is influenced by soil pH, salt concentration, temperature, redox potential, O$_2$ concentration, etc. Studies conducted on N$_2$O efflux under salt-affected conditions are summarized in Table 2. N$_2$O emissions from low salinity wetland soils (0.060 mg kg$^{-1}$) were reported higher as compared to high salinity wetland soils [26]. It is reported that the addition of salts up to a certain threshold may enhance the N$_2$O emissions and decrease thereafter [85]. The lower salinity level inhibits both the steps of nitrification (conversion of NH$_4^+$ to NO$_2^-$ and NO$_2^-$ to NO$_3^-$), however, the inhibition of NO$_2^-$ to NO$_3^-$ conversion is stronger than that of NH$_4^+$ to NO$_2^-$ causing higher NO$_2^-$ accumulation and enhanced N$_2$O emissions (Figure 3b) [85]. Li et al. [25] reported 110% higher N$_2$O emissions in slightly saline soil (1.0 dS m$^{-1}$) but 20% lower in moderate salinity (5 dS m$^{-1}$) soil as compared to non-saline soil (0.3 dS m$^{-1}$) under urea and ammonium sulfate fertilizer application. Ghosh et al. [30] observed that ammonical fertilizers, at low salinity, may enhance the N$_2$O emissions as compared to non-saline soil because that low salinity reduces the activity of N$_2$O reductase enzymes, therefore restricting the conversion of N$_2$O to N$_2$. In addition, low salinity can also serve as strong inhibition of nitrate oxidizers than ammonia oxidizers (Figure 3b) [36].

Soil moisture has a significant effect on N$_2$O emissions from saline soil. For instance, 4.7–37.6 and 5.0–15.3 times higher N$_2$O emissions at 100% soil moisture level than at 50 and 75% soil moisture levels, respectively, was reported by Li et al. [85], and it was mainly due to an increased rate of nitrification with increasing soil moisture [85]. Thapa et al. [86] reported that increasing salinity from 0.81 to 4.65 dS m$^{-1}$ increased N$_2$O flux at 90% of Water-filled pore space (WFPS) but reduced at 60% of WFPS in sulfate-dominated saline soil and it was due to that denitrifying bacteria performed more efficiently even at higher salinity levels because water addition reduces the adverse effect of salinity [86]. The quality of irrigation water plays important role in GHG emissions. Ma et al. [54] reported that saline water irrigation (8.01 dS m$^{-1}$) significantly reduced cumulative N$_2$O emissions than freshwater irrigation in calcareous soil. Wei et al. [87] reported that N$_2$O emissions decreased by 29.1 and 39.2% in 2 and 8 g L$^{-1}$ saline water irrigation, respectively, but increased by 58.3% in 5 g L$^{-1}$ treatments under N120 conditions as compared to freshwater irrigation. This investigation envisages that the sensitivity of N$_2$O production and consumption differed significantly with the degree of irrigation water salinities [87]. Nitrification is highly sensitive to salinity and lower N$_2$O emissions are expected with higher salinity levels. However, few researchers also reported the reverse trends [24,27,29].
### Table 2. Effect of soil salinity and sodicity on GHG emissions from saline ecosystems.

| References           | Systems and Study Locations                                                                 | Treatments Details                                                                 | Observation of the Study (GHG Emissions)                          | Key Findings/Reasoning                                                                 |
|----------------------|---------------------------------------------------------------------------------------------|----------------------------------------------------------------------------------|---------------------------------------------------------------------|---------------------------------------------------------------------------------------|
| **Field experiments**|                                                                                             |                                                                                  |                                                                     |                                                                                       |
| Li et al. [25]       | Field experiment at Kunshan field station, Suzhou, China.                                   | Nonsaline (S0), salinity-S1 (1 dS m\(^{-1}\)), salinity-S5 (5 dS m\(^{-1}\))   | \(\text{N}_2\text{O}\) emission increased by 89–110% at S1 and decreased by about 20% at S2. | Saline soils could be a potential source of \(\text{N}_2\text{O}\) emissions when cultivated. So, mitigation options should be explored. |
| Ma et al. [54]       | Long-term (2009–2018) field experiment in calcareous soil at the experimental station of Shihezi University, China. | N0 (no N) + SF (fresh water, 0.35 dS m\(^{-1}\)); N0 + SH (saline water, 8.04 dS m\(^{-1}\)); N360 (360 kg N ha\(^{-1}\)) + SF; N360 + SH | Irrigation with saline water inhibited \(\text{N}_2\text{O}\) emission, by 45.19% (N0) and 43.50% (N360) compared with irrigation with fresh water | Saline water irrigation altered community structures of denitrifying bacteria with nirK, nirS, and nosZ                                      |
| Capooci et al. [31]  | Field experiment at temperate salt marsh connected to the Atlantic Ocean.                   | Control (17 ppt); treatment (12.4–18.6 ppt)                                      | Soils subjected to low salinity had greater GHG emissions than control soils (17 ppt). Treatment soils had a 23% increase in GWP. | The storm surges can produce pulses of GHG emissions.                                   |
| Zhang et al. [88]    | GHG emissions from three rice (R1, R2, and R3) and maize (M1, M2, M3) fields with different salinity at Songyuan City of the Western Jilin Province, Northeast China. | R1 (pH: −7.6, EC: −0.16); R2 (pH: −8.6, EC: −0.27); R3 (pH: −9.1, EC: −0.41); M1 (pH: −7.34, EC: −0.10); M2 (pH: −7.76, EC: −0.19); M3 (pH: −8.43, EC: −0.25) | GWP of rice fields ranged 1070.0–1996.2 kg CO\(_2\) eq ha\(^{-1}\). Higher pH and salinity conditions consistently resulted in lower CO\(_2\), CH\(_4\), and \(\text{N}_2\text{O}\) emissions and CH\(_4\) uptake. |                                                                                                                                 |
| Poffenbarger et al. [89] | Metadata analysis based on secondary in-situ studies (31 nos.) of CH\(_4\) emissions from tidal marshes | Fresh (salinity < 0.5 ppt); Oligohaline (0.5–5.0 ppt); Mesohaline (5–18 ppt); Polyhaline (>18 ppt) | Oligohaline marshes had the highest and most variable CH\(_4\) emissions (150 ± 221 gm\(^{-2}\) yr\(^{-1}\)). Negligible CH\(_4\) emissions in polyhaline, and no significant difference between fresh and mesohaline marshes. | Need to estimate or monitor CH\(_4\) emissions in lower-salinity marshes.               |
| **Laboratory experiments** |                                                                                             |                                                                                  |                                                                     |                                                                                       |
| She et al. [90]      | Laboratory experiment with different textured soil.                                         | Sandy clay loam + S1 (0.10–1.0% soil salinity); Sandy loam + S1; Silty clay + S1 | Cumulative CO\(_2\) emissions in the coarse-textured (sandy clay loam and sandy loam) soils were more (206–231 and 176–204 mg CO\(_2\) kg\(^{-1}\)) affected by salinity than in the fine-textured (silty clay) soil. | Soil texture controlled the negative effect of salinity on C mineralization by regulating the soil microbial community composition. |
| Jia et al. [26]      | Low salinity wetland (LW-\(\text{Phragmites australis}\)) and High salinity wetland (HW-Suaeda sals) soils were collected and incubated. | LW soils (3.18 ppt salinity, 0.28 SAR); HW soils (13.30 ppt salinity, 0.03 SAR) | \(\text{N}_2\text{O}\) emissions were promoted in low salinity wetland (0.060 mg N\(_2\text{O}\) kg\(^{-1}\)) but significantly inhibited in high salinity wetlands (0.008 mg N\(_2\text{O}\) kg\(^{-1}\)) | Study suggests the complexity and uniqueness of \(\text{N}_2\text{O}\) emissions responses to nitrogen inputs related to the salinity levels. |
| References          | Systems and Study Locations                                                                 | Treatments Details                                                                 | Observation of the Study (GHG Emissions)                                                                 | Key Findings/Reasoning                                                                                                                                                                                                 |
|--------------------|---------------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------------------------|------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|
| Zhang et al. [91]   | Samples of saline-alkali soils were collected from four different locations in Yellow River Delta | Bare land soil (no vegetation) S0 (Control); S1 (1 mg g\(^{-1}\)); S3 (3 mg g\(^{-1}\)); S5 (5 mg g\(^{-1}\)) | CO\(_2\) emission ranged from 88.55 (S0)–51.77 (S3) mg CO\(_2\) kg\(^{-1}\) and N\(_2\)O 0.030 (S0)–0.012 (S3) mg N\(_2\)O kg\(^{-1}\). | The N\(_2\)O and CO\(_2\) emissions of were highest in herbage communities, intermediate in woody communities, and lowest in bare land under all treatments. The salinity effect on GHG emissions was stronger in soils with low salt levels. Higher GHG emissions at high soil moisture were found in all soils. |
| Ghosh et al. [30]    | Soil samples from three different locations within a salt affected agricultural land and incubated for 30 days. | S1 (0.44 dS m\(^{-1}\)); S2 (7.20 dS m\(^{-1}\)); S3 (4.55 dS m\(^{-1}\)) | The N\(_2\)O emissions significantly increased by 39.8% and 42.4% in S2 and S3, respectively. The addition of N significantly increased cumulative N\(_2\)O and CO\(_2\) emissions. | Saline-sodic soils can be a significant contributor to N\(_2\)O. Further, N fertilizer, irrigation, and precipitation may enhance GHG emissions. |
| Maucieri et al. [24] | Vertisol was collected from the experimental farm of University of Sydney and incubated for 30 days. | EC\(_{\text{aw}}\) (0.09 dS m\(^{-1}\)); EC\(_{\text{aw}}\) (5 dS m\(^{-1}\)); EC\(_{\text{aw}}\) (10 dS m\(^{-1}\)) | Saline water irrigation reduced CO\(_2\) emissions by 19% (5 dS m\(^{-1}\)) and 28% (10 dS m\(^{-1}\)). However, N\(_2\)O emissions increased by 60% with salinity | Salinity decreased CO\(_2\) and increased N\(_2\)O emission |
| Thapa et al. [86]    | Soil samples from Soil Health and Agriculture Research Extension (SHARE) farm were collected and incubated for 25 days. | EC<sub>c</sub> < 0.50 dS m\(^{-1}\) (60% WFPS); EC<sub>c</sub> 4.65 dS m\(^{-1}\) (60% WFPS); EC<sub>c</sub> 0.81 dS m\(^{-1}\) (90% WFPS); EC<sub>c</sub> 4.65 dS m\(^{-1}\) (90% WFPS); WFPS is a water-filled pore space | Relative decline in CO\(_2\) at higher EC<sub>c</sub> was smaller at 90% WFPS than at 90% WFPS. N\(_2\)O emission decreased by 45% at 60% WFPS and increased by 223% at 90% WFPS. | Higher soil moisture increased substrate availability, salt dilution, and enhance microbial activity, causing higher CO\(_2\) and N\(_2\)O emissions. |
| Reddy and Crohn [29] | Collected soil samples from abandoned field of Coachella Valley, California and incubated for 60 days | S3 (2.8 dS m\(^{-1}\)) (control); S15 (15.2 dS m\(^{-1}\)); S30 (30.6 dS m\(^{-1}\)) | Increased N\(_2\)O emission by 18–24% (at S15) and 34–67% (at S30), but decreased CO\(_2\) emissions | The use of active organic amendments to remediate salt-affected soils can prove to be beneficial in mitigating N\(_2\)O emission |
| Marton et al. [27]   | Soil samples were collected from tidal forests along the Altamaha, Ogeechee, and Satilla Rivers in southeast Georgia and incubated in the laboratory. | 0% (control); 2% saline water; 5% saline water | CH\(_4\) emission inhibited by 77% in the 2% and 89% in the 5% saline water treatment whereas CO\(_2\) generally increased with salinity, though exhibited a variable response between the three rivers. | Short-term salinity exposure enhanced anaerobic C mineralization, a decline in CH\(_4\) production, and a varied response in N\(_2\)O production |
| References        | Systems and Study Locations                                                                 | Treatments Details                                                                 | Observation of the Study (GHG Emissions)                                                                 | Key Findings/Reasoning                                                                 |
|-------------------|---------------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------|---------------------------------------------------------------------------------------------------------|----------------------------------------------------------------------------------------|
| Pattnaik et al. [92] | Soil samples from three locations (a) CRRI, Cuttack (alluvial soil), (b) Ernakulam, Kerala (acid sulphate saline soil (Pokkali)), and (c) Canning, West Bengal, (coastal saline soil) of India were collected and incubated for 35 days | Alluvial soil (0.35 dS m\(^{-1}\)) (control); Acid sulfate soil (5.01 dS m\(^{-1}\)); Coastal saline soil (17.23 dS m\(^{-1}\)) | CH\(_4\) production in non-saline alluvial soil was 630.86 ng CH\(_4\) g\(^{-1}\), and reduce remarkably in acid sulphate saline soil (12.97 ng CH\(_4\) g\(^{-1}\)), and coastal saline soil (142.36 CH\(_4\) g\(^{-1}\)) | High sulphate content of acid sulphate saline soil attributed to lower emission |
|                   |                                                                                             | Alluvial: (0.35 dS m\(^{-1}\)) (control); (4 dS m\(^{-1}\)); (8 dS m\(^{-1}\)); (16 dS m\(^{-1}\)); (20 dS m\(^{-1}\)) | Addition of salts to the non-saline alluvial soil at 4, 8, 16 and 20 dS m\(^{-1}\) progressively decreased CH\(_4\) production. | CH\(_4\) inhibition due to low microbial activities and soil microbial population including that of methanogens |
| Khatun et al. [28] | Pot study in net house at Bangladesh Agricultural University, Mymensingh, Bangladesh         | Control (100% NPK); control + 25 nM NaCl; control + 50 nM NaCl; control + 75 nM NaCl | Decreased yield scaled CH\(_4\) emission from 7.5% (25 nM NaCl) to 25% (75 nM NaCl)                  | Phosphogypsum and biochar with recommended fertilizers in saline soils could mitigate yield scaled CH\(_4\) emissions |
| Wei et al. [87]    | Collected soil samples from the greenhouse of Nanjing Vegetables Scientific Institute, China, and conducted pot experiments. | Freshwater (0.3 dS m\(^{-1}\) + N120 (120 kg N ha\(^{-1}\)); S1 (3.5 dS m\(^{-1}\)) + N120; S2 (8.1 dS m\(^{-1}\)) + N120; S3 (12.7 dS m\(^{-1}\)) + N120 | Irrigation with S1 water lowered N\(_2\)O emission and S2 enhanced emission by 58.3% | the effect degree of salinity on consumption and production of N\(_2\)O might vary among irrigation salinity ranges |
Figure 3. (a) Overview of the N$_2$O production pathways. (b) Inhibition of nitrification pathways due to soil salinity. The lower level of soil salinity slightly inhibits the activity of ammonia-oxidizing bacteria (AOB) and strongly inhibits the activity of nitrite-oxidizing bacteria (NOB) leading to the accumulation of NO$_2^-$ in soils and thereby high N$_2$O emissions. Adopted and modified from [18,25].

Reddy and Crohn [29] reported 0.004–0.007 mg N$_2$O kg$^{-1}$ soil at 2.8 dS m$^{-1}$, which increased by 18–24% and 34–87% at 15.2 and 30.6 dS m$^{-1}$, respectively (Table 2), mainly because of denitrification being the main process behind N$_2$O emissions. Similarly, Maucieri et al. [24] observed higher N$_2$O emissions from increased irrigation water salinity. Marton et al. [27] also reported higher N$_2$O emissions from tidal forest soil in southeast Georgia irrigated with high levels of irrigation water salinities. The possible reasons behind the higher N$_2$O emissions at higher salinity levels were (a) decoupling of either denitrification and nitrification processes [93] and (b) the higher salinity levels have increased the sulfate reduction leading to a higher concentration of H$_2$S which is popularly known as the inhibitor for N$_2$O reduction [94,95]. Another possible reason is that high salinity suppresses the activity of N$_2$O reductase leading to N$_2$O accumulation due to denitrification under a saline environment [73,96].

Overall, it can be concluded that both soil and irrigation water salinity significantly affect the N$_2$O emissions from soils. Usually, the higher the levels of salts, the lower the N$_2$O emissions [26,66]. Although, soil salinity is a limiting factor affecting the nitrification and denitrification processes by changing the microbial growth and activity as well as the physical and chemical properties of soil. Beside this moisture is the key factor influencing nitrification and denitrification in salt-affected soils [65]. Denitrification had a significantly positive relationship with soil moisture and it increased with an increase in soil moisture when the soil water content was less than 27.03% and decreased with an increase in soil moisture when the soil water content was more than 27.03% [66].

There are several mechanisms behind this. Firstly, high salinity stress enhances the ammonification and inhibited the nitrification process resulting in reduced NO$_3^-$ and a rise in NH$_4^+$ concentration in soils, which restricted the concentration of substrate for the denitrification, the main process responsible for N$_2$O production [66,97]. Laura [98] reported that high salinity stress completely inhibited the process of nitrification due to a decrease in the nitrifying community in the soils. Further higher ionic strength at high salinity levels can adsorb the exchangeable NH$_4^+$ ions [99]. Secondly, the diversity of denitrifying bacteria is reduced at higher salinity stress [100,101], which may lower down microbial-mediated denitrification process limiting the N$_2$O production. Finally, at higher salinity, the dissimilatory nitrate reduction to ammonium (DNRA) and denitrification processes compete with each other for a common substrate, i.e., NO$_3^-$, and thereby the rate of nitrification is limited due to the unavailability of NO$_3^-$ [102]. In contrast, several
researchers reported higher N₂O emissions with increased salinity stress mainly due to the reduction of N₂O to N₂ conversion and higher N mineralization [24,36]. Jia et al. [26] reported an increased rate of denitrification with the addition of salts up to 1–5 ppt but decreased with 10 ppt salts level. Zhou et al. [103] reported a higher N₂O to N₂ ratio with high salinity treatments. Altogether, it is concluded that the relationship between the N₂O production and salinity levels significantly depends on various processes of production and consumption/reduction of N₂O under salinity stress, which is highly variable with variations in levels of salinity, moisture, soil pH, and concentration of NO₃⁻ and NH₄⁺, etc. in soils.

4.2. Methane Emissions

Methane is produced by methanogenic bacteria/archaea (methanogens) as the end product of organic matter decomposition under anaerobic conditions [104,105]. Under an anaerobic atmosphere, methanogens utilize the methanogenic substrates (methanol, formate, acetate, and CO₂) produced due to organic matter decomposition by a range of heterotrophic organisms (Figure 4) [113]. CH₄ is consumed by methanotrophs as a source of energy and carbon and they can grow both in aerobic and anaerobic environments [10,15]. Several factors such as pH, salt concentration, redox potential, soil organic matter, microbial community, etc. affect the CH₄ emissions from soils [10,106]. Soil pH and salt concentration significantly affect the methane emissions from soils and generally, it is inversely related to the salinity/sodicity. Usually, CH₄ emissions from normal/non-saline soil are higher as compared to saline and sodic soil [28,107]. CH₄ emissions reported from the saline soils of different ecosystems are given in Table 2. Sun et al. [108] reported 71% higher CH₄ emissions from non-saline inland soils than that from coastal soil in a meta-analysis. Datta et al. [109] reported similar results from a field experiment in India. They reported higher CH₄ emissions (279.79–378.20 kg CH₄ ha⁻¹) from non-saline soil as compared to saline soil (123.87–170.46 kg CH₄ ha⁻¹) under similar management practices. The findings of Sun et al. [108] and Datta et al. [109] suggest that higher concentrations of exchangeable cations (Na⁺, K⁺, Ca²⁺, and Mg²⁺) were the main reason behind the lower CH₄ emissions from saline soils. High soil salinity suppresses the activity of methanogens resulting in lower CH₄ production and subsequent emissions [110–113].

Recently, Khatun et al. [28] reported an inverse relationship between salinity stress and CH₄ emissions and CH₄ emissions were 6.6, 6.1, 5.6, and 4.9 g CH₄ pot⁻¹ season⁻¹ with 0, 25, 50, and 75 mM NaCl stress, respectively. In China, Zhang et al. [88] investigated CH₄ emissions from rice soil reduced significantly by about 16 and 39 in 0.27 and 0.41% salinity levels, respectively, as compared to the 0.16% salinity (Table 2). The higher salinity levels inhibited the CH₄ oxidation potential of methanotrophs resulting in higher CH₄ emissions over lower salinity levels [88]. Martin et al. [27] collected tidal forest soils from three different locations and incubated them in the laboratory with 0, 2, and 5% salinity levels of irrigation water and reported that CH₄ emissions decreased from 0.51 to 0.07 mg CH₄ kg⁻¹ h⁻¹ by increasing salinity from 0 to 5% (Table 2).

Pattnaik et al. [92] reported significantly higher (3.78 mg CH₄ kg⁻¹ soil) mean CH₄ emissions from non-saline alluvial soil as compared to acid sulfate soil (0.02 mg CH₄ kg⁻¹ soil). The major reason behind this was the high sulfate content of acid sulfate soil which enhances the population of sulfate-reducing bacteria (SRB), and these SRB compete with the methanogens for common substrates (Figure 4) [110]. CH₄ emissions were significantly lower in the coastal saline soil (0.25 mg CH₄ kg⁻¹ soil) as compared to non-saline alluvial soil (3.78 mg CH₄ kg⁻¹ soil), which was mainly due to the higher salt content of coastal saline soils. Further, the non-saline alluvial soil was incubated with different levels of salinity, and it was reported that inhibition of the CH₄ production was directly proportional to the salinity levels. The average CH₄ emissions were reduced by 55% at 4 dS m⁻¹ salinity and almost inhibited at the 20 dS m⁻¹ salinity. Maucieri et al. [24] and Zhang et al. [88] reported that a higher level of soil salinity reduces the CH₄ uptake by methanotrophs. Zhang et al. [88] studied the CH₄ uptake/emissions from three maize
fields (M1, M2, and M3) having different salinity and sodicity levels. The cumulative CH$_4$ uptake was 0.77 kg CH$_4$ ha$^{-1}$ from M1 (pH: $-7.34$, EC: $-0.10$), and it was reduced by 16% and 24% in the case of M2 (pH: $-7.76$, EC: $-0.19$) and M3 (pH: $-8.43$, EC: $-0.25$) soils. Similarly, Maucieri et al. [24] collected soil samples in pots and studied the CH$_4$ uptake/emissions by irrigating with three types of saline water (0.5/0.9, 5.0, and 10 dS m$^{-1}$). Total CH$_4$ uptake/emissions was 0.07, 0.07, and 0.06 mg CH$_4$ kg$^{-1}$ soil in 0.9, 5.0, and 10 dS m$^{-1}$, respectively. Earlier Poffenbarger et al. [89] did the metadata analysis for CH$_4$ emissions from tidal marshes [fresh (salinity < 0.5 ppt), oligohaline (0.5–5.0 ppt), mesohaline (5–18 ppt), and polyhaline (>18 ppt)] and reported that CH$_4$ emissions from the fresh, oligohaline, mesohaline, and polyhaline marshes were 419, 1500, 164 and 11.2 kg ha$^{-1}$ year$^{-1}$. Generally, CH$_4$ emissions decreased with increased salinity of tidal marshes, however, there is a need to monitor or estimate the CH$_4$ emissions from oligohaline marshes.

**Figure 4.** Schematic diagram showing the process of CH$_4$ production, mechanism of reduction of CH$_4$ production with the application of gypsum and phosphogypsum through competition with the sulfate-reducing bacteria, Modified from [114,115].

### 4.3. Carbon Dioxide Emissions

Both salinity and sodicity induce a significant impact on CO$_2$ emissions from soils and usually, CO$_2$ emissions have an inverse relation with salinity [90,111] and a positive relationship with soil temperature. Recently, Yu et al. [116] studied the process of CO$_2$ emissions under different levels of soil salinity and temperature and observed a positive correlation between CO$_2$ emissions, soil salinity, and temperature [116]. Soil temperature significantly affects the microbial population prevailing in the saline soil (Figure 5). At higher temperatures gram-positive bacterial and fungal populations dominated in the saline soil and these microbial populations effectively decomposed soil organic carbon pool into CO$_2$ [117]. She et al. [90] studied the effect of salinity levels on CO$_2$ emissions and found that under similar salinity levels (0.10–1.0%), the highest CO$_2$ emissions were
reported from sandy clay loam soil (206–231 mg CO$_2$ kg$^{-1}$ day$^{-1}$) followed by sandy loam and lowest from silty clay. Zhang et al. [91] collected soils from four different soils having different salinity and vegetation types, i.e., bare soil (EC: 14.84 mS cm$^{-1}$), T. chinensis community (EC: 10.46 mS cm$^{-1}$), S. salsa community (EC: 5.18 mS cm$^{-1}$), and P. australis community (EC: 2.47 mS cm$^{-1}$). They incubated these soils with four salinity levels as control, 1 mg g$^{-1}$, 3 mg g$^{-1}$, and 5 mg g$^{-1}$. CO$_2$ emissions from the bare land were the lowest (51.76 to 88.55 mg kg$^{-1}$ soil). Whereas CO$_2$ emissions, from the soils with communities of T. chinensis, S. salsa, and P. australis: were from 231.46 to 282.25, 400.39 to 504.33, and 391.27 to 518.46 mg kg$^{-1}$ soil, respectively (Table 2). CO$_2$ emissions from the same soils with different salinity levels were decreased with increased salinity. It was observed that the CO$_2$ emissions from these soils were positively correlated with available labile soil carbon [91]. Degradation of above and below-ground biomass enhanced the labile carbon which results in higher CO$_2$ emissions from salt-affected soil with vegetation cover than bare salt-effected soil. Maucieri et al. [24] incubated the Vertisol soil in the laboratory adjusting irrigation water salinity to 0.09, 5.0, and 10.0 dS m$^{-1}$ using NaCl and studied GHG emissions. They reported that with increasing salinity, CO$_2$ emissions decreased by 19% (5 dS m$^{-1}$) and 28% (10 dS m$^{-1}$) as compared to control (0.09 dS m$^{-1}$). Reddy and Crohn [29] collected soil samples from Coachella Valley, California having three different levels of EC$_e$ (2.8, 15.2, and 30.6 dS m$^{-1}$) and measured CO$_2$ emissions in an incubation experiment and reported 70–371 mg CO$_2$ kg$^{-1}$ soil in 2.8 dS m$^{-1}$, 38–259 mg CO$_2$ kg$^{-1}$ soil in 15.2 dS m$^{-1}$, and 12.47–187 mg CO$_2$ kg$^{-1}$ soil in 30.6 dS m$^{-1}$ salinity level.

**Figure 5.** Mechanism of carbon dioxide production in saline soils.
5. Impact of Soil Amendments on GHG Emissions from Salt-Affected Soils

Reclamation and sustainable management of salt-affected soils for economic production of crops is a global challenge and expected climate change has further aggravated the task. Several management practices including gypsum, phosphogypsum, organic matter, biochar, vermicompost, etc. were evaluated and promoted for the management of these soils (Table 3). These amendments are being used to improve salt-affected soils for agricultural performance. Besides, they also have an impact on soil microbial activities and thus may enhance or reduce GHG emissions. Therefore, the impact of various reclamation technologies/materials on GHG emissions needs a global scientific investigation.

Table 3. Different amendments/materials used for the management of salt-affected soils and mitigation of GHG emissions.

| S. No. | Amendment Details                          | References     |
|--------|--------------------------------------------|----------------|
| 01     | Gypsum/phosphogypsum                       | [83,118,119]   |
| 02     | Biochar                                    | [81,114,115]   |
| 03     | Humic acid                                 | [118]          |
| 04     | Rice straw compost                         | [120]          |
| 05     | Cow manure                                 | [81]           |
| 06     | Deep tillage                               | [121]          |
| 07     | Vermicompost                               | [121]          |
| 08     | Azolla application                         | [122]          |
| 09     | Cyanobacteria                              | [123]          |
| 10     | Jatropha curcas                            | [124]          |
| 11     | Sesbania green manure                      | [125]          |
| 12     | Nitrification inhibitors (3,4-Dimethylpyrazole phosphate) | [126] |

5.1. Impact of Gypsum and Phosphogypsum Application on GHG Emissions

Gypsum and phosphogypsum are used for centuries to reclaim alkali soils. Once gypsum dissociates into calcium and sulfur, calcium has the greatest attraction for the soil particle displacing sodium and helps flocculate (aggregate) the soils to improve soil structure. Besides, it might also affect the soil microbes involved in GHG emissions. Several literatures are available on the gypsum application and reclamation of alkali soils. However, very selective studies are available that assessed the effect of these soil corrections on GHG emissions. Some studies [28,83,118] studied the CH$_4$ emissions in the rice ecosystems and observed its mitigation in reclaimed soils. Denier van der Gon and Neue, [127] reported 55–70% lower CH$_4$ emissions from gypsum amended rice fields and it is most likely due to the inhibition of the methanogenesis by the sulfate reductase bacteria (SRB) (Table 4). Gypsum application enhance SO$_4^{2-}$ concentration, which gives rise to the competition of SRB with the methanogens for common substrate (H$_2$, CO$_2$, acetate) that otherwise be used by methanogens [127,128] as SRB has a high affinity for H$_2$ and acetate as compared to methanogens [129]. This inhibition of methanogenesis by SRB is incomplete and a considerable amount of CH$_4$ emissions still occurred [127]. The gypsum and phosphogypsum improve water infiltration through betterment in structure, thereby enhancing soil redox potential (less negative redox potential) and mitigating CH$_4$ emissions from saline/sodic soils [28]. The rate of gypsum application also plays a significant role in CH$_4$ emissions. Theint et al. [83] reported that CH$_4$ emissions from alkali soils significantly increased with the application of a lower dose (0.5 t ha$^{-1}$) of gypsum as rhizospheric exudates provide a sufficient amount of Organic C for methanogenesis. However, a higher dose (2 t ha$^{-1}$) reduced the CH$_4$ emissions by lowering the soil pH and increased sulfate concentration. The CH$_4$ mitigation is generally higher with the higher dose of gypsum.
application, and it is because of higher competition between SRB and methanogens at a higher rate. One mole of \( \text{SO}_4^{2-} \) is required for the reduction of one mole of \( \text{CH}_4 \) [130]. Overall, it can be concluded that gypsum is the best reclamation agent which can be used as a mitigating agent for \( \text{CH}_4 \) emissions from paddy cultivation in both sodic and non-sodic soils and mitigation is higher at a higher rate of application. Park et al. [131] reported about 60% \( \text{CH}_4 \) mitigation from paddy soils with the application of 8 MG ha\(^{-1}\) by-product gypsum fertilizer (BGF) application. Similarly, Ali et al. [132] reported 18–23% \( \text{CH}_4 \) mitigation from the coastal paddy soils with the application of silicate slag (150 kg ha\(^{-1}\)). Both BGF and silicate slag had high free iron oxide and \( \text{SO}_4^{2-} \) content which acted as electron acceptors. Sun et al. [118] explored the potential of gypsum and humic acid on \( \text{CH}_4 \) and \( \text{N}_2\text{O} \) emissions from coastal saline soils and recorded mitigation of 19.36% \( \text{CH}_4 \) emissions and 9.43% \( \text{N}_2\text{O} \) emissions in gypsum amended N fertilized soils (Table 4). The application of humic acid in coastal saline soil served as electron acceptors, which result in higher \( \text{CH}_4 \) emissions as compared to no application of humic acid in soils. Further, the application of humic acid enhanced the soil redox potential which stimulate the higher \( \text{N}_2\text{O} \) fluxes from the soils [118].

### Table 4. Impact of different amendments practices on GHG emissions from salt-affected soils.

| References          | Experiment Type       | Treatment's Detail                                                                 | Observation (GHG Emissions)                                                                 | Key Findings and Reasoning                                                                 |
|---------------------|-----------------------|----------------------------------------------------------------------------------|-------------------------------------------------------------------------------------------|------------------------------------------------------------------------------------------|
| Khatun et al. [28]  | Pot experiment        | Initial soil pH = 7.8, EC = 5.6 dS m\(^{-1}\), OC = 1.48%                        | 25 nM salinity; 25 nM + phosphogypsum (P); 50 mM; 50 nM + P; 75 nM; 75 nM + P;            | Phospho-gypsum and biochar mitigate \( \text{CH}_4 \) emissions due to improved soil redox potential (Eh), increased \( \text{SO}_4^{2-} \) and decreased soil EC. |
| Sun et al. [118]    | Field experiment      | Field experiment growing rice conducted in Jiangsu Province, China               | N1 (300 kg N ha\(^{-1}\)); N1 + humic acid; N1 + gypsum; N1 + humic acid + gypsum           | Humic acid and gypsum application with N300 kg N ha\(^{-1}\) is the better management for coastal saline soils of China to mitigate \( \text{CH}_4 \) emission. |
| Park et al. [131]   | Field experiment with rice |                                                                 | No by-product gypsum fertilizer (BGF); BGF (2 Mg ha\(^{-1}\)); BGF (4 Mg ha\(^{-1}\)); BGF (8 Mg ha\(^{-1}\)) | \( \text{CH}_4 \) flux decreased with increasing level of BGF, and BGF (8 Mg ha\(^{-1}\)) reduced it by 60.6% compared to control. |
| Ali et al. [132]    | Field experiment with rice in upland soil |                                                                 | Urea (250 kg ha\(^{-1}\)); Urea + Phosphogypsum (90 kg ha\(^{-1}\)); Urea + silicate slag (150 kg ha\(^{-1}\)) | Silicate slag and phosphogypsum reduced \( \text{CH}_4 \) emission by 18.0–23.5% and 14.7–18.6%, respectively. |
| Denier van der Gon and Neuw. [127] | Field experiment with rice |                                                                 | Urea (165 kg N ha\(^{-1}\)); Urea + gypsum (6.60 t ha\(^{-1}\)) | The \( \text{CH}_4 \) emissions from gypsum amended plots were reduced by 55–70% compared to non-amended plots. |
| Sun et al. [133]    | Field experiment with rice |                                                                 | N1 (300 kg N ha\(^{-1}\)); N1 + 20 t biochar ha\(^{-1}\); N1 + 40 t biochar ha\(^{-1}\) | Biochar amendment increased \( \text{N}_2\text{O} \) emissions by 13.7–38.1% and had no significant effects on \( \text{CH}_4 \) emissions. Thus, long-term observations are needed to evaluate the environmental impacts of biochar and N fertilizers. |
Table 4. Cont.

| References                  | Experiment Type                  | Treatment’s Detail                                                                 | Observation (GHG Emissions) | Key Findings and Reasoning |
|-----------------------------|----------------------------------|-----------------------------------------------------------------------------------|-------------------------------|-----------------------------|
| Maucieri et al. [24]        | 30 days incubation experiment    | Control; Biochar                                                                  | Biochar amendment to saline soil decrease CH$_4$ uptake (8.8%), CO$_2$ (11.9%), and N$_2$O (9.8%) emissions | Biochar amendment to soils mitigates GHG emissions where CO$_2$ and N$_2$O are driven by soil rewetting events. |
| Datta et al. [109]          | Rice experiment in irrigated saline soils of Gadakujang (a fishing hamlet) of coastal Odisha, India | Prilled urea (40 kg N ha$^{-1}$); Sesbania green manure (5 Mg ha$^{-1}$) + prilled urea (20 kg N ha$^{-1}$); Ipomoea lacunose (5 Mg ha$^{-1}$) + prilled urea (28 kg N ha$^{-1}$) | Sesbania and Ipomoea lacunose green manure reduced CH$_4$ emission by 23.2 and 29.9%. | Locally available Ipomoea lacunose green manure can use CH$_4$ mitigation and yield enhancement from the coastal saline rice ecosystems |
| Denier van der Gon and Neue, [127] | Field experiment with rice GM (S. Rostrata: 20 t ha$^{-1}$) + urea (30 kg urea ha$^{-1}$); GM + urea + gypsum (6.60 t ha$^{-1}$) | Green manure addition enhances CH$_4$ emissions by 10 times than that of urea application alone, further gypsum addition reduced CH$_4$ emission by about 71.1%. | Database for CH$_4$ emissions mitigation from rice grown on high-sulfate containing soils |
| Chen et al. [134]           | Field experiment was conducted in saline-sodic soils in the upper Yellow River basin, Northwest China | Organic fertilizer (CK), sheep manure (FYM), lignite bioorganic fertilizer (LBF1) (1.5 t ha$^{-1}$) LBF2 (3 t ha$^{-1}$), LBF3 (4.5 t ha$^{-1}$), and LBF4 (7.5 t ha$^{-1}$) | LBF treatments decreased CH$_4$ and CO$_2$, and increasing N$_2$O emissions beyond 3 t ha$^{-1}$ application rate. FYM acted as a CH$_4$ source, and LBF2 and LBF3 treatments acted as CH$_4$ sinks. | The application of lignite bioorganic fertilizer at 3.0-4.5 t ha$^{-1}$ is appropriate for GHG mitigation in saline-sodic farmlands. |
| Zheng et al. [135]          | Microcosm experiments of 80 days incubation Interaction of salinity (0 and 1.2% salt) with biochar | 5–10 times higher N$_2$O emissions occurred from saline soils than that from non-saline soils. Aged biochar decreased N$_2$O emissions and increased CO$_2$ emissions in saline soils. | Aged biochar could be a better option for mitigation of N$_2$O emissions from saline soils. |
| Li et al. [126]             | Field experiment with rice crop Nonsaline (NS) soil; NS soil + DMPP (0.8% w/w of N); low saline (LS) soil; LS soil + DMPP; high saline (HS) soil; HS soil + DMPP | The nitrification inhibitor DMPP (3,4-dimethyl pyrazole phosphate) reduced cumulative N$_2$O emissions by 61% in non-saline soil and by 75% in low saline soil | DMPP offsets low salinity-induced high N$_2$O emissions by inhibiting ammonia oxidation. |

5.2. Organic Amendments and GHG Emissions

Organic manure, green manure, biochar, compost, etc. are commonly used for the management of salt-affected soils and biochar has a great potential for N$_2$O mitigation [24,136]. The application of fresh organic matter and biochar improves soil’s physical, chemical, and biological properties and can either enhance or mitigate N$_2$O emissions [29,136]. Application of fresh organic matter (crop residue, manure, FYM) increases cumulative N$_2$O emissions due to enhanced nitrogen mineralization [137]. However, biochar application in saline soils inhibits the nitrification process through adsorption of substrate, i.e., NH$_3$/NH$_4^+$ and resulting in a lowering of N$_2$O emissions [24,138]. Biochar at the rate of 1% of total N in saline-alkali soil reduced nirK and nirS gene copies of denitrifiers bacteria and resulted in low N$_2$O emissions [115]. The age of the biochar is also important in GHG mitigation, aged biochar further decreases N$_2$O emissions from saline soils. Therefore, aged biochar could be a better option for the mitigation of N$_2$O emissions from these soils.

Substituting inorganic fertilizer with organic matter in optimum portion can be useful in maintaining SOC in agricultural soil along with CH$_4$ mitigation [15,139,140]. The application of biochar significantly enhanced the community structure and abundance of methanotrophs which reduces the net CH$_4$ emissions from biochar-treated soils [136,141]. The application of FYM in rice grown in saline soil significantly enhances the popula-
tion of methanotrophs [142]. Similarly, the use of FYM along with pyrite in alkaline paddy soil enhances the methanotroph population and CH$_4$ oxidation thereby reducing seasonal CH$_4$ emissions [142]. Wang et al. [143] reported three to six times higher methanotroph population and lower CH$_4$ emissions with the use of biochar along with steel slag. Nguyen et al. [81] observed that cow manure addition to salt-affected soil enhanced CH$_4$ emissions by 801%, however, the addition of biochar to cow manure amended soil reduced CH$_4$ emissions from 28 to 680%. The application of cow manure alone enhanced the population of methanogens leading to significantly higher CH$_4$ emissions. While application of biochar along with cow manure enhanced the methanotrophs population and thereby improved the net balance of methanogens (CH$_4$ production) and methanotrophs (CH$_4$ consumption) resulting in lowering the CH$_4$ fluxes from biochar + cow manure amended soils [10,81].

*Sesbania aculeate* and *Ipomoea lacunose* green manure reduced CH$_4$ emissions by 23.15 and 29.89%, respectively, as compared to urea application during the wet season (Table 4). However, this green manure enhanced CH$_4$ emissions by 382.68, and 300.57%, respectively, during the dry season [109] because the higher temperature in the dry season accelerates the process of fresh green manure. Maucieri et al. [24] reported 10% lower cumulative CO$_2$ emissions and 12% lower N$_2$O emissions from biochar amended soil as compared to without biochar amended soil. Supparattanapan et al. [144] conducted a field experiment in a saline patch (14.2 dS m$^{-1}$) and outside saline patch (4.7 dS m$^{-1}$) of a single field and reported that the addition of rice straw and cow manure enhanced the CH$_4$ emissions from both saline and outside saline patch over control. However, increased CH$_4$ emissions were higher from the outside saline patch (153–161%) as compared to the saline patch (33–19.5%). Overall, it can be concluded that biochar can be used as the best organic amendment for mitigation of N$_2$O and CH$_4$ emissions from both normal soils as well as salt-affected soils.

5.3. Other Interventions for GHG Mitigation from Saline-Sodic Soils

Several other materials were also tested for the GHG mitigation potential from the salt-affected Soils. Li et al. [126] investigated the role of 3,4-Dimethylpyrazole phosphate (DMPP) a new nitrification inhibitor in reducing N$_2$O emissions from saline soil and reported that the application of DMPP in non-saline, low saline, and high saline soil significantly reduced N$_2$O emissions by 61.19, 74.94, and 48.82%, respectively, over non DMPP treatment (Table 4). DMPP application reduced the NO$_2$-N accumulation and suppressed nitrifier denitrification processes and causes a lower N$_2$O emissions [126]. Sun et al. [108] reported that the application of an acid chemical (trade name—Hekang) at the rate of 22.5 kg ha$^{-1}$ along with urea application at the rate of 300 kg N ha$^{-1}$ did not reduce N$_2$O emissions from the saline soil but reduced the yield scaled emissions.

6. Future Research Directions

Based on the literature reviewed following thrust areas are identified which required future research attention for the low carbon/GHG emissions and sustainable crop production from the salt-affected lands/soils.

1. The Impacts of excess salts and high pH on GHG emissions from salt-affected soils are well documented. However, impacts of the individual ion toxicity on microbial population, enzymatic activities, and GHG production processes required further investigation.
2. Mostly, studies are conducted in the pot and laboratory under controlled conditions. However, in real field conditions the emissions may be affected by several other parameters, therefore, how salinity and sodicity in actual field conditions affect the soil GHG emissions needs further investigation.
3. How the other parameters such as soil carbon and nitrogen level, soil moisture, redox potential, precipitation, temperature, cyclones, etc. affect the seasonal variation of GHG emission from salt-affected soils before and after reclamation needs systematic investigation.
4. Systematic investigations are needed to understand and quantify the effect of different amendments and reclamation technologies such as gypsum, phosphogypsum, organic manure, green manure, biochar, etc. on GHG emissions from these soils to develop the low carbon emission reclamation technologies for the management of salt-affected soils.

7. Conclusions

Salinity and sodicity not only affect the soil’s physicochemical properties but also significantly affects the CH$_4$, N$_2$O, and CO$_2$ emissions from the soil. The production of GHG in the soil is mainly governed by the microbial-mediated processes in which several organisms are involved. Salinity and sodicity have detrimental impacts on the microbial population of nitrifying, denitrifying, methanogens, methanotrophs, etc., and enzymatic activities involved in GHG production and consumption. The magnitude of its impact depends on the level of salinity and sodicity. Microbial population and soil enzyme activities are generally decreased with increasing soil salinity and sodicity which restrict the C and N mineralization and thereby GHG emissions. Generally, CH$_4$ production and emissions from soils decrease with increasing soil salinity which is mainly due to the inhibition of methanogens activity. Similarly, N$_2$O emissions also decreased with increasing salinity due to strong inhibition of both steps of nitrification. However, N$_2$O emission enhanced at lower levels of salinity as compared to non-saline soils. Reclamation of salt-affected soils using various amendments normalizes the soil pH and reduces soil salinity which is favorable for the microbial population and can enhance GHG emissions from soils. However, reclamation of salt-affected soils using gypsum and phosphogypsum reduces the CH$_4$ emissions from soils mainly through the competition due to sulfate-reducing bacteria for the common substrates. The rate of gypsum application has a greater impact on CH$_4$ mitigation from salt-affected soils. Similarly, biochar amendments to soil reduce both CH$_4$ and N$_2$O emissions and mitigation is higher with aged biochar. The application of fresh organic matter and FYM may enhance GHG emissions. Although, the amendments and reclamation technologies are used to make crop cultivation possible from these soils. However, systematic investigations are needed for studying the impacts of various reclamation technologies on GHG emissions so that low carbon emissions reclamation technologies can be promoted in the policies for the reclamation of salt-affected soils, and the carbon sequestration potential of these soils can be explored.

Author Contributions: Conceptualization, R.K.F.; methodology, R.K.F. and S.K.M.; formal analysis, R.K.F. and S.K.M.; writing—original draft preparation, R.K.F., S.K.M. and D.S.; writing—review and editing, A.K., R.K.Y., P.C.S. and H.P. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Acknowledgments: The authors are grateful to the Prioritization, Monitoring and Evaluation (PME) cell of ICAR-CSSRI, Karnal, for approval of the manuscript (Review article/21/2021). We are thankful to the Editor and the three anonymous reviewers for their constructive and insightful comments which improved the quality of the manuscript a lot.

Conflicts of Interest: The authors declare no conflict of interest.

References
1. Ding, Z.; Koriem, M.A.; Ibrahim, S.M.; Antar, A.S.; Ewis, M.A.; He, Z.; Kheir, A.M.S. Seawater Intrusion Impacts on Groundwater and Soil Quality in the Northern Part of the Nile Delta, Egypt. Environ. Earth Sci. 2020, 79, 313. [CrossRef]
2. FAOSTAT. Food and Agriculture Organization of the United Nations, Rome, Italy. 2020. Available online: http://faostat.fao.org/ (accessed on 4 April 2022).
3. Amini, S.; Ghadiri, H.; Chen, C.; Marschner, P. Salt-Affected Soils, Reclamation, Carbon Dynamics, and Biochar: A Review. J. Soils Sediments 2016, 16, 939–953. [CrossRef]

4. Snoussi, M.; Ouchani, T.; Niazi, S. Vulnerability Assessment of the Impact of Sea-Level Rise and Flooding on the Moroccan Coast: The Case of the Mediterranean Eastern Zone. Estuar. Coast. Shelf Sci. 2008, 77, 206–213. [CrossRef]

5. IPPC IPCC (Intergovermental Panel on Climate Change). Synthesis Report 5; IPPC IPCC (Intergovermental Panel on Climate Change): Geneva, Switzerland, 2014.

6. Nouri, H.; Chavoshi Borjjeni, S.; Nirola, R.; Hassanli, A.; Beecham, S.; Alaghamd, S.; Saint, C.; Mulcahy, D. Application of Green Remediation on Soil Salinity Treatment: A Review on Halophytoremediation. Process Saf. Environ. Prog. 2017, 107, 94–107. [CrossRef]

van Weert, F.; van der Gun, J.; Reckman, J. Global Overview of Saline Groundwater Occurrence and Genesis (Report Number: GP 2009-1). Ultr. IGRAC—U. N. Int. Groundw. Resour. Assess. Cent. 2009, 1–32.

7. Shani, U.; Dudley, L.M. Field Studies of Crop Response to Water and Salt Stress. Soil Sci. Soc. Am. J. 2001, 65, 1522–1528. [CrossRef]

Tubiello, F.N.; Salvatore, M.; Rossi, S.; Ferrara, A.; Fitton, N.; Smith, P. The FAOSTAT Database of Greenhouse Gas Emissions from Agriculture. Environ. Res. Lett. 2013, 8. [CrossRef]

8. Snoussi, M.; Ouchani, T.; Niazi, S. Vulnerability Assessment of the Impact of Sea-Level Rise and Flooding on the Moroccan Coast: The Case of the Mediterranean Eastern Zone. Estuar. Coast. Shelf Sci. 2008, 77, 206–213. [CrossRef]

9. Tubiello, F.N.; Salvatore, M.; Rossi, S.; Ferrara, A.; Fitton, N.; Smith, P. The FAOSTAT Database of Greenhouse Gas Emissions from Agriculture. Environ. Res. Lett. 2013, 8. [CrossRef]

10. Malyan, S.K.; Bhatia, A.; Kumar, A.; Gupta, D.K.; Singh, R.; Kumar, S.S.; Tomer, R.; Kumar, O.; Jain, N. Methane Production, Oxidation and Mitigation: A Mechanistic Understanding and Comprehensive Evaluation of Influencing Factors. Sci. Total Environ. 2016, 572, 874–896. [CrossRef] [PubMed]

11. Fagodiya, R.K.; Pathak, H.; Kumar, A.; Bhatia, A.; Jain, N. Global Temperature Change Potential of Nitrogen Use in Agriculture: A 50-Year Assessment. Sci. Rep. 2017, 7, 1–8. [CrossRef] [PubMed]

12. Oertel, C.; Matschullat, J.; Zurba, K.; Zimmermann, F.; Erasmi, S. Greenhouse Gas Emissions from Soils—A Review. Chemie Der Erdte—Geochim. 2016, 76, 327–352. [CrossRef]

13. Conrad, R. Microbial Ecology of Methanogens and Methanotrophs. Adv. Agron. 2007.

14. Conrad, R. Methane Production in Soil Environments—Anaerobic Biogeochemistry and Microbial Life between Flooding and Desiccation. Microorganisms 2020, 8, 881. [CrossRef]

15. Fagodiya, R.K.; Pathak, H.; Bhatia, A.; Tomer, R.; Harit, R.C.; Jain, N.; Bhowmik, A.; Kaushik, R. Mitigation of Yield-Scaled Greenhouse Gas Emissions from Irrigated Rice through Azolla, Blue-Green Algae, and Plant Growth–Promoting Bacteria. Environ. Sci. Pollut. Res. 2021. [CrossRef] [PubMed]

16. Yemadje, P.L.; Chevallier, T.; Guibert, H.; Bertrand, I.; Bernoux, M. Wetting-Drying Cycles Do Not Increase Organic Carbon and Nitrogen Mineralization in Soils with Straw Amendment. Geoderma 2017, 304, 68–75. [CrossRef]

17. Khoi, C.M.; Guong, V.T.; Merckx, R. Predicting the Release of Mineral Nitrogen from Hypersaline Pond Sediments Used for Brine Remediation on Soil Salinity Treatment: A Review on Halophytoremediation. Process Saf. Environ. Prog. 2017, 107, 94–107. [CrossRef]

18. Wang, S.; Tang, J.; Li, Z.; Liu, Y.; Zhou, Z.; Wang, J.; Qu, Y.; Dai, Z. Carbon Mineralization under Different Saline-Alkali Stress Conditions in Paddy Fields of Northeast China. Sustainability 2020, 12, 2921. [CrossRef]

19. Usmani, D.; Lal, R. Soil Emission of Nitrous Oxide and Its Mitigation. Soil Emiss. Nitrous Oxide its Mitig. 2013, 1–28. [CrossRef]

20. Wang, S.; Tang, J.; Li, Z.; Liu, Y.; Zhou, Z.; Wang, J.; Qu, Y.; Dai, Z. Carbon Mineralization under Different Saline-Alkali Stress Conditions in Paddy Fields of Northeast China. Sustainability 2020, 12, 2921. [CrossRef]

21. Yemadje, P.L.; Chevallier, T.; Guibert, H.; Bertrand, I.; Bernoux, M. Wetting-Drying Cycles Do Not Increase Organic Carbon and Nitrogen Mineralization in Soils with Straw Amendment. Geoderma 2017, 304, 68–75. [CrossRef]

22. Wang, S.; Tang, J.; Li, Z.; Liu, Y.; Zhou, Z.; Wang, J.; Qu, Y.; Dai, Z. Carbon Mineralization under Different Saline-Alkali Stress Conditions in Paddy Fields of Northeast China. Sustainability 2020, 12, 2921. [CrossRef]

23. Ichida, K.; Usmani, D.; Lal, R. Influence of Salinity and Water Content on Soil Microorganisms. Int. Soil Water Conserv. Res. 2013, 3, 316–323. [CrossRef]

24. Ichida, K.; Usmani, D.; Lal, R. Influence of Salinity and Water Content on Soil Microorganisms. Int. Soil Water Conserv. Res. 2013, 3, 316–323. [CrossRef]

25. Ichida, K.; Usmani, D.; Lal, R. Influence of Salinity and Water Content on Soil Microorganisms. Int. Soil Water Conserv. Res. 2013, 3, 316–323. [CrossRef]

26. Ichida, K.; Usmani, D.; Lal, R. Influence of Salinity and Water Content on Soil Microorganisms. Int. Soil Water Conserv. Res. 2013, 3, 316–323. [CrossRef]
32. Rengasamy, P. Soil Salinity and Sodicity. In Growing Crops with Reclaimed Wastewater; Siro Publishing: Clayton, VIC, Australia, 2006; pp. 125–138.

33. Rengasamy, P. Soil Processes Affecting Crop Production in Salt-Affected Soils. Funct. Plant Biol. 2010, 37, 613–620. [CrossRef]

34. Richards, L. Diagnosis and Improvement of Saline and Alkaline Soils. Soil Sci. Soc. Am. J. 1954, 18, 348. [CrossRef]

35. Rengasamy, P. Transient Salinity and Subsoil Constraints to Dryland Farming in Australian Sodic Soils: An Overview. Aust. J. Exp. Agric. 2002, 42, 351–361. [CrossRef]

36. Zhou, M.; Butterbach-Bahl, K.; Vereecken, H.; Brüggemann, N. A Meta-Analysis of Soil Salinization Effects on Nitrogen Pools, Cycles and Fluxes in Coastal Ecosystems. Glob. Chang. Biol. 2017, 23, 1338–1352. [CrossRef] [PubMed]

37. Qadir, M.; Quillierou, E.; Nangia, V.; Murtaza, G.; Singh, M.; Thomas, R.J.; Drechsel, P.; Noble, A.D. Economics of Salt-Induced Land Degradation and Restoration. Nat. Resour. Forum 2014, 38, 282–295. [CrossRef]

38. Shahid, S.A.; Zaman, M.; Heng, L. Soil Salinity: Historical Perspectives and a World Overview of the Problem. Guidel. Salin. Assess. Mitig. Adapt. Using Nucl. Relat. Tech. 2018, 43–53. [CrossRef]

39. Issanova, G.T.; Abuduwaili, J.; Mamutov, Z.U.; Kaldybaev, A.A.; Saparov, G.A.; Bazarbaeva, T.A. Saline Soils and Identification of Salt Accumulation Provinces in Kazakhstan. Arid Ecosyst. 2017, 7, 243–250. [CrossRef]

40. Toderich, K.; Ismail, S.; Massino, I.; Wilhelm, M.; Yusupov, S.; Kuliev, T. Extent of Salt-Affected Land in Central Asia: Biosaline Agriculture and Utilization of the Salt-Affected Resources; KIER Working Papers 648; Kyoto University, Institute of Economic Research: Kyoto, Japan, 2018.

41. NRI/SA Salt Affected Soils, National Remote Sensing Agency. Department of Space, Government of India, Hyderabad; National Remote Sensing Agency, Department of Space, Government of India: Hyderabad, India, 1997.

42. Singh, G.; Bundela, D.S.; Sethi, M.; Lal, K.; Kamra, S.K. Remote Sensing and Geographic Information System for Appraisal of Salt-Affected Soils in India. J. Environ. Qual. 2010, 39, 5–15. [CrossRef]

43. FAO; ITPS. Status of the World’s Soil Resources. Total Environ. 2017, 595, 472–485. [CrossRef]

44. Siddikee, M.A.; Tipayno, S.C.; Kim, K.; Chung, J.; Sa, T. Influence of Varying Degree of Salinity-Sodicity Stress on Enzyme Activities and Bacterial Populations of Coastal Soils of Yellow Sea, South Korea. J. Microbiol. Biotechnol. 2011, 21, 341–346. [CrossRef]

45. Behera, P.; Mahapatra, S.; Mohapatra, M.; Kim, J.Y.; Adhya, T.K.; Raina, V.; Suar, M.; Pattnaik, A.K.; Rastogi, G. Salinity and Macrophyte Drive the Biogeography of the Sedimentary Bacterial Communities in a Brackish Water Tropical Coastal Lagoon. Sci. Total. Environ. 2017, 595, 472–485. [CrossRef]

46. Franklin, R.B.; Morrissey, E.M.; Morina, J.C. Changes in Abundance and Community Structure of Nitrate-Reducing Bacteria in a Created Freshwater Wetland. Appl. Microbiol. Biotechnol. 2007, 73, 189–197. [CrossRef]

47. Li, C.; Lei, J.; Zhao, Y.; Xu, X.; Li, S. Effect of Saline Water Irrigation on Soil Development and Plant Growth in the Taklimakan Desert Highway Shelterbelt. Soil Tillage Res. 2015, 146, 99–107. [CrossRef]

48. Bhullar, R.S.; Mavi, M.S.; Choudhary, O.P. Adverse Impact of Sodicity on Soil Functions Can Be Alleviated through Addition of Rice Straw Biochar. Commun. Soil Sci. Plant Anal. 2019, 50, 2369–2383. [CrossRef]

49. Larsen, L.; Moseman, S.; Santoro, A.E.; Hopfensperger, K.; Burgin, A. A Complex-Systems Approach to Predicting Effects of Sea Level Rise and Nitrogen Loading on Nitrogen Cycling in Coastal Wetland. Eco-DAS VIII 2010, 67–92. [CrossRef]

50. Giblin, A.E.; Weston, N.B.; Banta, G.T.; Tucker; J; Hopkinson, C.S. The Effects of Salinity on Nitrogen Losses from an Oligohaline Estuarine Sediment. Estuaries Coasts 2010, 33, 1054–1068. [CrossRef]

51. Rysgaard, S.; Thastum, P.; Dalsgaard, T.; Christensen, P.B.; Sloth, N.P. Effects of Salinity on Nitrogen Fixation and Denitrification in Danish Estuarine Sediments. Estuaries 1999, 22, 21–30. [CrossRef]

52. Franklin, R.B.; Morrissey, E.M.; Morina, J.C. Changes in Abundance and Community Structure of Nitrate-Reducing Bacteria along a Salinity Gradient in Tidal Wetlands. Pedobiologia 2017, 60, 21–26. [CrossRef]

53. Oren, A. Anaerobic Degradation of Organic Compounds at High Salt Concentrations. Antonie Van Leeuwenhoek 1988, 54, 267–277. [CrossRef]

54. Ma, L.; Guo, H.; Min, W. Nitrous Oxide Emission and Denitrifier Bacteria Communities in Calcareous Soil as Affected by Drip Irrigation with Saline Water. Appl. Soil Ecol. 2019, 143, 222–233. [CrossRef]

55. Marchant, H.K.; Lavik, G.; Halttappel, M.; Kuypers, M.M.M. The Fate of Nitrate in Intertidal Permeable Sediments. PLoS ONE 2014, 9, e104517. [CrossRef] [PubMed]

56. Tong, C.; She, C.X.; Yang, P.; Jin, Y.F.; Huang, J.F. Weak Correlation Between Methane Production and Abundance of Methanogens Across Three Brackish Marsh Zones in the Min River Estuary, China. Estuaries Coasts 2015, 38, 1872–1884. [CrossRef]

57. Scott, J.T.; McCarthy, M.J.; Gardner, W.S.; Doyle, R.D. Denitrification, Dissimilatory Nitrate Reduction to Ammonium, and Nitrogen Fixation along a Nitrate Concentration Gradient in a Created Freshwater Wetland. Biogeochemistry 2008, 87, 99–111. [CrossRef]

58. Zhang, Y.; Chen, L.; Dai, T.; Tian, J.; Wen, D. The Influence of Salinity on the Abundance, Transcriptional Activity, and Diversity of AOA and AOB in an Estuarine Sediment: A Microcosm Study. Appl. Microbiol. Biotechnol. 2015, 99, 9825–9833. [CrossRef]

59. Wang, H.; Gilbert, J.A.; Zhu, Y.; Yang, X. Salinity Is a Key Factor Driving the Nitrogen Cycling in the Mangrove Sediment. Sci. Total. Environ. 2018, 631–632, 1342–1349. [CrossRef] [PubMed]

60. Li, M.; Gu, J.D. Community Structure and Transcript Responses of Anammox Bacteria, AOA, and AOB in Mangrove Sediment Microcosms Amended with Ammonium and Nitrate. Appl. Microbiol. Biotechnol. 2013, 97, 9859–9874. [CrossRef] [PubMed]
62. Guo, H.; Ma, L.; Liang, Y.; Hou, Z.; Min, W. Response of Ammonia-Oxidizing Bacteria and Archaea to Long-Term Saline Water Irrigation in Alluvial Grey Desert Soils. *Sci. Rep.* 2020, 10, 489. [CrossRef] [PubMed]

63. Kaushik, A.; Sethi, V. Salinity Effects on Nitrifying and Free Diazotrophic Bacterial Populations in the Rhizosphere of Rice. *Bull. Natl. Inst. Ecol.* 2005, 15, 139–144.

64. Magallães, C.M.; Joye, S.B.; Moreira, R.M.; Wiebe, W.J.; Bordalo, A.A. Effect of Salinity and Inorganic Nitrogen Concentrations on Nitrification and Denitrfication Rates in Intertidal Sediments and Rocky Biofilms of the Douro River Estuary, Portugal. *Water Res.* 2005, 39, 1783–1794. [CrossRef] [PubMed]

65. Liu, B.; Zhao, W.; Wen, Z.; Yang, Y.; Chang, X.; Yang, Q.; Meng, Y.Y.; Liu, C. Mechanisms and Feedbacks for Evapotranspiration-Induced Salt Accumulation and Precipitation in an Arid Wetland of China. *J. Hydrol.* 2019, 568, 403–415. [CrossRef]

66. Meng, Y.; He, Z.; Liu, B.; Chen, L.; Lin, P.; Luo, W. Soil Salinity and Moisture Control the Processes of Soil Nitrification and Denitrification in a Riparian Wetlands in an Extremely Arid Regions in Northwestern China. *Water* 2020, 12, 2815. [CrossRef]

67. Fagodiya, R.K.; Kumar, A.; Kumari, S.; Medhi, K.; Shabnam, A.A. Role of Nitrogen and Its Agricultural Management in Changing Environment. In *Contaminants in Agriculture*; Springer: Berlin/Heidelberg, Germany, 2020; pp. 247–270, ISBN 9783030415525.

68. Braker, G.; Conrad, R. Diversity, Structure, and Size of N₂O-Producing Microbial Communities in Soils—What Matters for Their Functioning? *Adv. Appl. Microbiol.* 2011, 73, 33–70. [PubMed]

69. Wang, X.; Wang, S.; Shi, G.; Wang, W.; Zhu, H. Factors Driving the Distribution and Role of AOA and AOB in Phragmites Communis Rhizosphere in Riparian Zone. *J. Basic Microbiol.* 2019, 59, 425–436. [CrossRef] [PubMed]

70. Zhang, L.; Song, L.; Zhang, L.; Shao, H.; Chen, X.; Yan, K. Seasonal Dynamics in Nitrous Oxide Emissions under Different Types of Vegetation in Saline-Alkaline Soils of the Yellow River Delta, China and Implications for Eco-Restoring Coastal Wetland. *Ecol. Eng.* 2013, 61, 82–89. [CrossRef]

71. Shao, X.; Zhao, L.; Sheng, X.; Wu, M. Effects of Influent Salinity on Water Purification and Greenhouse Gas Emissions in Lab-Scale Constructed Wetlands. *Environ. Sci. Pollut. Res.* 2020, 27, 21487–21496. [CrossRef]

72. Fiedler, D.J.; Clay, D.E.; Joshi, D.R.; Engel, A.; Marzano, S.Y.; Jakubowski, D.; Bhattacharai, D.; Reese, C.L.; Bruggeman, S.A.; Clay, S.A. CO₂ and N₂O Emissions and Microbial Community Structure from Fields That Include Salt-Affected Soils. *J. Environ. Qual.* 2021, 50, 567–579. [CrossRef]

73. Dalal, R.C.; Wang, W.; Robertson, G.P.; Parton, W.J. Nitrous Oxide Emission from Australian Agricultural Lands and Mitigation Options: A Review. *Aust. J. Soil Res.* 2003, 41, 165–195. [CrossRef]

74. Wang, D.; Chen, Z.; Sun, W.; Hu, B.; Xu, S. Methane and Nitrous Oxide Concentration and Emission Flux of Yangtze Delta Plain River Net. *Sci. China Ser. B Chem.* 2009, 52, 652–661. [CrossRef]

75. Costa, K.C.; Leigh, J.A. Metabolic Versatility in Methanogens. *Curr. Opin. Biotechnol.* 2014, 29, 70–75. [CrossRef]

76. Ollivier, B.; Caumette, P.; Garcia, J.L.; Mah, R.A. Anaerobic Bacteria from Hypersaline Environments. *Microbiol. Rev.* 1994, 58, 27–38. [CrossRef]

77. Scholten, J.C.M.; Joye, S.B.; Hollibaugh, J.T.; Murrell, J.C. Molecular Analysis of the Sulfate Reducing and Archaeal Community in a Meromictic Soda Lake (Mono Lake, California) by Targeting 16S rRNA, MerA, ApSₐ, and DsrAB Genes. *Microb. Ecol.* 2005, 50, 29–39. [CrossRef]

78. Bébut, B.M.; Hoehler, T.M.; Thamdrup, B.; Albert, D.; Carpenter, S.P.; Hogan, M.; Turk, K.; Des Marais, D.J. Methane Production by Microbial Mats under Low Sulphate Concentrations. *Geobiology* 2004, 2, 87–96. [CrossRef]

79. Smith, J.M.; Green, S.J.; Kelley, C.A.; Prufert-Bebout, L.; Bebout, B.M. Shifts in Methanogen Community Structure and Function Associated with Long-Term Manipulation of Sulfate and Salinity in a Hypersaline Microbial Mat. *Environ. Microbiol.* 2008, 10, 386–394. [CrossRef] [PubMed]

80. Heyer, J.; Berger, U.; Hardt, M.; Dunfield, P.F. Methylohalobius Crimeensis Gen. Nov, Sp. Nov, a Moderately Halophilic, Methanotrophic Bacterium Isolated from Hypersaline Lakes of Crimea. *Int. J. Syst. Evol. Microbiol.* 2005, 55, 1817–1826. [CrossRef] [PubMed]

81. Nguyen, B.T.; Trinh, N.N.; Bach, Q.V. Methane Emissions and Associated Microbial Activities from Paddy Salt-Affected Soil as Influenced by Biochar and Cow Manure Addition. *Appl. Soil Ecol.* 2020, 152, 103531. [CrossRef]

82. Xiao, K.; Guo, C.; Maspolim, Y.; Zhou, Y.; Ng, W.J. The Role of Methanogens in Acetic Acid Production under Different Salinity Conditions. *Chemosphere* 2016, 161, 53–60. [CrossRef]

83. Thein, E.E.; Suzuki, S.; Ono, E.; Bellingrath-Kimura, S.D. Influence of Different Rates of Gypsum Application on Methane Emission from Saline Soil Related with Rice Growth and Rhizosphere Exudation. *Caten* 2015, 133, 467–473. [CrossRef]

84. Weston, N.B.; Vile, M.A.; Neubauer, S.C.; Velinsky, D.J. Accelerated Microbial Organic Matter Mineralization Following Salt-Water Intrusion into Tidal Freshwater Marsh Soils. *Biogeochemistry* 2011, 102, 135–151. [CrossRef]

85. Li, Y.; Xu, J.; Liu, B.; Wang, H.; Qi, Z.; Wei, Q.; Liao, L.; Liu, S. Enhanced N₂O Production Induced by Soil Salinity at a Specific Range. *Int. J. Environ. Res. Public Health* 2020, 17, 5169. [CrossRef]

86. THAPA, R.; CHATTERJEE, A.; WICK, A.; BUTCHER, K. Carbon Dioxide and Nitrous Oxide Emissions from Naturally Occurring Sulfate-Based Saline Soils at Different Moisture Contents. *Plosphere* 2017, 27, 868–876. [CrossRef]

87. Wei, Q.; Xu, J.; Liao, L.; Li, Y.; Wang, H.; Rahim, S.F. Water Salinity Should Be Reduced for Irrigation to Minimize Its Risk of Increased Soil N₂O Emissions. *Int. J. Environ. Res. Public Health* 2018, 15, 2114. [CrossRef]

88. Zhang, H.; Tang, J.; Liang, S.; Li, Z.; Yang, P.; Wang, J.; Wang, S. The Emissions of Carbon Dioxide, Methane, and Nitrous Oxide during Winter without Cultivation in Local Saline-Alkali Rice and Maize Fields in Northeast China. *Sustainability* 2017, 9, 1916. [CrossRef]
23 of 25

89. Poffenbarger, H.J.; Needelman, B.A.; Megonigal, J.P. Salinity Influence on Methane Emissions from Tidal Marshes. *Wetlands* 2011, 31, 831–842. [CrossRef]

90. She, R.; Yu, Y.; Ge, C.; Yao, H. Soil Texture Alters the Impact of Salinity on Carbon Mineralization. *Agronomy* 2021, 11, 128. [CrossRef]

91. Zhao, L.; Song, L.; Wang, B.; Shao, H.; Zhang, L.; Qin, X. Co-Effects of Salinity and Moisture on CO₂ and N₂O Emissions of Laboratory-Incubated Salt-Affected Soils from Different Vegetation Types. *Geoderma* 2018, 332, 109–120. [CrossRef]

92. Pattnaik, P.; Mishra, S.R.; Bharati, K.; Mohanty, S.R.; Sethunathan, N.; Adhya, T.K. Influence of Salinity on Methanogenesis and Associated Microflora in Tropical Rice Soils. *Microbiol. Res.* 2000, 155, 215–220. [CrossRef]

93. Low, A.P.; Stark, J.M.; Dudley, L.M. Effects of Soil Osmotic Potential on Nitrification, Ammonification, N-Assimilation, and Nitrous Oxide Production. *Soil Sci.* 1997, 162, 16–27. [CrossRef]

94. Jäntti, H.; Aalto, S.L.; Paerl, H.W. Inhibition of Ferrous Iron and Hydrogen Sulfide on Nitrate Reduction in the Sediments of an Estuary Experiencing Hypoxia. *Estuaries and Coasts* 2021, 44, 1–12. [CrossRef]

95. Serensen, J.; Tiedje, J.M.; Firestone, R.B. Inhibition by Sulphide of Nitric and Nitrous Oxide Reduction by Denitrifying Pseudomonas Fluorescens. *Appl. Environ. Microbiol.* 1980, 39, 105–108. [CrossRef] [PubMed]

96. Han, Z.; Dong, J.; Shen, Z.; Mou, R.; Zhou, Y.; Chen, X.; Fu, X.; Yang, C. Nitrogen Removal of Anaerobically Digested Swine Wastewater by Pilot-SCALE Tidal Flow Constructed Wetland Based on in-Situ Biological Regeneration of Zeolite. *Chemosphere* 2019, 217, 364–373. [CrossRef]

97. Jia, W.; Sun, X.; Gao, Y.; Yang, Y.; Yang, C. Fe-Modified Biochar Enhances Microbial Nitrogen Removal Capability of Constructed Wetland. *Sci. Total Environ.* 2020, 740, 139534. [CrossRef]

98. Laura, R.D. Salinity and Nitrogen Mineralization in Soil. *Soil Bio. Biochem.* 1977, 9, 333–336. [CrossRef]

99. Weston, N.B.; Giblin, A.E.; Banta, G.T.; Hopkinson, C.S.; Tucker, J. The Effects of Varying Salinity on Ammonium Exchange in Estuarine Sediments of the Parker River, Massachusetts. *Estuaries and Coasts* 2010, 33, 985–1003. [CrossRef]

100. Xu, S.; Fu, X.; Ma, S.; Bai, Z.; Xiao, R.; Li, Y.; Zhuang, G. Mitigating Nitrous Oxide Emissions from Tea Field Soil Using Bioaugmentation with a Trichoderma Viride Biofertilizer. *Sci. World J.* 2014, 2014, 793752. [CrossRef]

101. Ilgrande, C.; Leroy, B.; Wattiez, R.; Vlaeminck, S.E.; Boon, N.; Clauwaert, P. Metabolic and Proteomic Responses to Salinity in Synthetic Nitrifying Communities of *Nitrosomonas* spp. And *Nitrobacter* spp. *Front. Microbiol.* 2018, 9, 2914. [CrossRef]

102. Babbin, A.R.; Tamasi, T.; Dumit, D.; Weber, L.; Rodriguez, M.V.I.; Schwartz, S.L.; Armenteros, M.; Wankel, S.D.; Aprill, A. Discovery and Quantification of Anaerobic Nitrogen Metabolism among Oxigenated Tropical Cuban Stony Corals. *ISME J.* 2021, 15, 1222–1235. [CrossRef]

103. Zhao, Y.; Xia, Y.; Li, B.; Yan, X. Influence of Environmental Factors on Net N₂ and N₂O Production in Sediment of Freshwater Rivers. *Environ. Sci. Pollut. Res.* 2014, 21, 9973–9982. [CrossRef] [PubMed]

104. Malyan, S.K.; Singh, O.; Kumar, A.; Anand, G.; Singh, R.; Singh, S.; Yu, Z.; Kumar, J.; Fagodiya, R.K.; Kumar, A. Greenhouse Gases Trade-Off from Ponds: An Overview of Emission Process and Their Driving Factors. *Water* 2022, 14, 970. [CrossRef]

105. Malyan, S.K.; Kumar, S.S.; Singh, A.; Kumar, O.; Gupta, D.K.; Yadav, A.N.; Fagodiya, R.K.; Khan, S.A.; Kumar, A. Understanding Methanogens, Methanotrophs, and Methane Emission in Rice Ecosystem. In *Microbiomes and the Global Climate Change*; Springer: Singapore, 2021; pp. 205–224, ISBN 9789813345089.

106. Hussain, S.; Peng, S.; Fahad, S.; Khalq, A.; Huang, J.; Cui, K.; Nie, L. Rice Management Interventions to Mitigate Greenhouse Gas Emissions: A Review. *Environ. Sci. Pollut. Res.* 2015, 22, 3342–3360. [CrossRef]

107. Vo, T.B.T.; Wassmann, R.; Tirol-Padre, A.; Cao, V.P.; MacDonald, B.; Espaldon, M.V.O.; Sander, B.O. Methane Emission from Rice Cultivation in Different Agro-Ecological Zones of the Mekong River Delta: Seasonal Patterns and Emission Factors for Baseline Water Management. *Soil Sci. Plant Nutr.* 2018, 64, 47–58. [CrossRef]

108. Sun, M.; Zhang, H.; Dong, J.; Gao, F.; Li, X.; Zhang, R. A Comparison of CH₄ Emissions from Coastal and Inland Rice Paddy Soils in China. *Catena* 2018, 170, 365–373. [CrossRef]

109. Das, S.; Ganguly, D.; De, T.K. Microbial Methane Production-Oxidation Profile in the Soil of Mangrove and Paddy Fields of West Bengal, India. *Geomicrobiol. J.* 2020, 38, 220–230. [CrossRef]

110. Saifullah; Dahlawi, S.; Naeem, A.; Rengel, Z.; Naidu, R. Biochar Application for the Remediation of Salt-Affected Soils: Challenges and Opportunities. *Sci. Total Environ.* 2018, 625, 320–335. [CrossRef]

111. Shi, Y.; Liu, X.; Zhang, Q. Effects of Combined Biochar and Organic Fertilizer on Nitrous Oxide Fluxes and the Related Nitrifier and Denitrifier Communities in a Saline-Alkali Soil. *Sci. Total Environ.* 2019, 686, 199–211. [CrossRef]

112. Jia, W.; Sun, X.; Gao, Y.; Yang, Y.; Yang, C. Fe-Modified Biochar Enhances Microbial Nitrogen Removal Capability of Constructed Wetland. *Sci. Total Environ.* 2020, 740, 139534. [CrossRef]

113. Das, S.; Ganguly, D.; De, T.K. Microbial Methane Production-Oxidation Profile in the Soil of Mangrove and Paddy Fields of West Bengal, India. *Geomicrobiol. J.* 2020, 38, 220–230. [CrossRef]

114. Saifullah; Dahlawi, S.; Naeem, A.; Rengel, Z.; Naidu, R. Biochar Application for the Remediation of Salt-Affected Soils: Challenges and Opportunities. *Sci. Total Environ.* 2018, 625, 320–335. [CrossRef]

115. Shi, Y.; Liu, X.; Zhang, Q. Effects of Combined Biochar and Organic Fertilizer on Nitrous Oxide Fluxes and the Related Nitrifier and Denitrifier Communities in a Saline-Alkali Soil. *Sci. Total Environ.* 2019, 686, 199–211. [CrossRef]

116. Yu, Y.; Li, X.; Zhao, C.; Zheng, N.; Jia, H.; Yao, H. Soil Salinity Changes the Temperature Sensitivity of Soil Carbon Dioxide and Nitrous Oxide Emissions. *Catena* 2020, 195, 104912. [CrossRef]
117. Toor, M.; Kumar, S.S.; Malyan, S.K.; Bishnoi, N.R.; Mathimani, T.; Rajendran, K.; Pugazhendhi, A. An Overview on Bioethanol Production from Lignocellulosic Feedstocks. *Chemosphere* 2020, 242, 125080. [CrossRef]

118. Sun, L.; Ma, Y.; Liu, Y.; Li, J.; Deng, J.; Rao, X.; Zhang, Y. The Combined Effects of Nitrogen Fertilizer, Humic Acid, and Gypsum on Yield-Scaled Greenhouse Gas Emissions from a Coastal Saline Rice Field. *Environ. Sci. Pollut. Res.* 2019, 26, 19502–19511. [CrossRef]

119. Ali, M.K.; Ahmad, W.; Malhi, S.S.; Atta, B.M.; Ghafoor, A.; Zia, M.H. Potential of Carbon Dioxide Biosequestration of Saline-Sodic Soils during Amelioration under Rice-Wheat Land Use. *Commun. Soil Sci. Plant Anal.* 2015, 44, 2625–2635. [CrossRef]

120. Kim, Y.J.; Choo, B.K.; Cho, J.Y. Effect of Gypsum and Rice Straw Compost Application on Improvements of Soil Quality during Desalination of Reclaimed Coastal Tideland Soils: Ten Years of Long-Term Experiments. *Catena* 2017, 156, 131–138. [CrossRef]

121. Ding, Z.; Kheir, A.M.S.; Ali, O.A.M.; Hafez, E.M.; ElShamey, E.A.; Zhou, Z.; Wang, B.; Lin, X.; Ge, Y.; Fahmy, A.E.; et al. A Vermicompost and Deep Tillage System to Improve Saline-Sodic Soil Quality and Wheat Productivity. *J. Environ. Manag.* 2021, 277. [CrossRef] [PubMed]

122. Abou Hussien, E.; Ahmed, B.; Elbaalawy, A. Efficiency of Azolla and Biochar Application on Rice (*Oryza Sativa*) L. Productivity in Salt-Affected Soil. *Egypt. J. Soil Sci.* 2020, 60, 277–288. [CrossRef]

123. Li, H.; Zhao, Q.; Huang, H. Current States and Challenges of Salt-Affected Soil Remediation by Cyanobacteria. *Sci. Total Environ.* 2019, 669, 258–272. [CrossRef] [PubMed]

124. Singh, K.; Singh, B.; Tuli, R. Sodic Soil Reclamation Potential of Jatropha Curcas: A Long-Term Study. *Ecol. Eng.* 2013, 58, 434–440. [CrossRef]

125. Sarwar, G.; Malik, M.A.; Tahir, M.A.; Aftab, M.; Manzoor, M.Z.; Zafar, A. Comparative Efficiency of Compost, Farmyard Manure and Sesbania Green Manure to Produce Rice-Wheat Crops under Salt Stressed Environmental Conditions. *J. Pure Appl. Agric.* 2020, 5, 33–42.

126. Li, Y.; Xu, J.; Liu, X.; Qi, Z.; Wang, H.; Li, Y.; Liao, L. Nitrification Inhibitor DMPP Offsets the Increase in N2O Emission Induced by Soil Salinity. *Biol. Fertil. Soils* 2020, 56, 1211–1217. [CrossRef]

127. Denier van der Gon, H.A.C.; Neue, H.U. Impact of Gypsum Application on the Methane Emission from a Wetland Rice Field. *Global Biogeochem. Cycles* 1994, 8, 127–134. [CrossRef]

128. Epule, E.T.; Peng, C.; Mafany, N.M. Methane Emissions from Paddy Rice Fields: Strategies towards Achieving A Win-Win Sustainability Scenario between Rice Production and Methane Emission Reduction. *J. Sustain. Dev.* 2011, 4. [CrossRef]

129. Gauci, V.; Matthews, E.; Dime, N.; Waller, B.; Koch, D.; Granberg, G.; Vile, M. Sulfur Pollution Suppression of the Wetland Methane Source in the 20th and 21st Centuries. *Proc. Natl. Acad. Sci. USA* 2004, 101, 12583–12587. [CrossRef] [PubMed]

130. Denier van der Gon, H.A.; van Bodegom, P.M.; Wassmann, R.; Lantin, R.S.; Metra-Corton, T.M. Sulfate-Containing Amendments to Reduce Methane Emissions from Rice Fields: Mechanisms, Effectiveness and Costs. *Mitig. Adapt. Strateg. Glob. Chang.* 2001, 6, 71–89. [CrossRef]

131. Park, J.-H.; Sonn, Y.-K.; Kong, M.-S.; Zhang, Y.-S.; Park, S.-J.; Won, J.-G.; Lee, S.-H.; Seo, D.-H.; Park, S.-D.; Kim, J.-E. Effect of By-Product Gypsum Fertilizer on Methane Gas Emissions and Rice Productivity in Paddy Field. *Korean J. Soil Sci. Fertil.* 2016, 49, 30–35. [CrossRef]

132. Ali, M.; Farouque, M.; Haque, M.; Ul Kabir, A. Influence of Soil Amendments on Mitigating Methane Emissions and Sustaining Rice Productivity in Paddy Soils Ecosystems of Bangladesh. *J. Environ. Sci. Nat. Resour.* 2012, 5, 179–185. [CrossRef]

133. Sun, L.; Deng, J.; Fan, C.; Li, J.; Liu, Y. Combined Effects of Nitrogen Fertilizer and Biochar on Greenhouse Gas Emissions and Net Ecosystem Economic Budget from a Coastal Saline Rice Field in Southeastern China. *Environ. Sci. Pollut. Res.* 2020, 27, 17013–17022. [CrossRef]

134. Chen, Z.; Huang, G.; Li, Y.; Zhang, X.; Xiong, Y.; Huang, Q.; Jin, S. Effects of the Lignite Bioorganic Fertilizer on Greenhouse Gas Emissions and Pathways of Nitrogen and Carbon Cycling in Saline-Sodic Farmlands at Northwest China. *J. Clean. Prod.* 2022, 56, 139, 74–79. [CrossRef]

135. Zheng, N.; Yu, Y.; Li, Y.; Ge, C.; Chapman, S.J.; Yao, H. Can Aged Biochar Offset Soil Greenhouse Gas Emissions from Crop Residue Amendments in Saline and Non-Saline Soils under Laboratory Conditions? *Sci. Total Environ.* 2022, 806, 151256. [CrossRef]

136. Malyan, S.K.; Kumar, S.S.; Fagodiya, R.K.; Ghosh, P.; Kumar, A.; Singh, R.; Singh, L. Biochar for Environmental Sustainability in the Energy-Water-Agroecosystem Nexus. *Renew. Sustain. Energy Rev.* 2021, 149, 111379. [CrossRef]

137. Yu, Y.; Zhao, C.; Zheng, N.; Jia, H.; Yao, H. Interactive Effects of Soil Texture and Salinity on Nitrous Oxide Emissions Following Crop Residue Amendment. *Geoderma* 2019, 337, 1146–1154. [CrossRef]

138. Zhu, H.; Yang, J.; Yao, R.; Wang, X.; Xie, W.; Zhu, W.; Liu, X.; Cao, Y.; Tao, J. Interactive Effects of Soil Amendments (Biochar and Gypsum) and Salinity on Ammonia Volatilization in Coastal Saline Soil. *Catena* 2020, 190. [CrossRef]

139. Bhattacharya, S.S.; Kim, K.H.; Das, S.; Uchimiy, M.; Jeon, B.H.; Kwon, E.; Szulejko, J.E. A Review on the Role of Organic Inputs in Maintaining the Soil Carbon Pool of the Terrestrial Ecosystem. *J. Environ. Manag.* 2016, 167, 214–227. [CrossRef] [PubMed]

140. Nan, Q.; Wang, C.; Wang, H.; Yi, Q.; Wu, W. Mitigating Methane Emission via Annual Biochar Amendment Pyrolyzed with Rice Straw from the Same Paddy Field. *Sci. Total Environ.* 2020, 746, 141351. [CrossRef] [PubMed]

141. Singh, J.S.; Pandey, V.C.; Singh, D.P.; Singh, R.P. Influence of Pyrite and Farmyard Manure on Population Dynamics of Soil Methanotroph and Rice Yield in Saline Rain-Fed Paddy Field. *Agric. Ecosyst. Environ.* 2010, 139, 74–79. [CrossRef]
143. Wang, M.; Wang, C.; Lan, X.; Abid, A.A.; Xu, X.; Singla, A.; Sardans, J.; Llusià, J.; Peñuelas, J.; Wang, W. Coupled Steel Slag and Biochar Amendment Correlated with Higher Methanotrophic Abundance and Lower CH$_4$ Emission in Subtropical Paddies. *Environ. Geochem. Health* **2020**, *42*, 483–497. [CrossRef]

144. Supparattanapan, S.; Saenjan, P.; Quantin, C.; Maeght, J.L.; Grünberger, O. Salinity and Organic Amendment Effects on Methane Emission from a Rain-Fed Saline Paddy Field. *Soil Sci. Plant Nutr.* **2009**, *55*, 142–149. [CrossRef]