EXPLORING MITIGATION OPTION OF GREENHOUSE GAS (GHG) EMISSIONS UNDER WATER MANAGEMENTS AND SOIL AMELIORATIONS FROM MINERAL SOIL AND PEAT SOIL IN INDONESIA

February 2016

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CHIBA UNIVERSITY

(千葉大学学位申請論文）

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A thesis submitted to Graduate School of Horticulture, Chiba University, in partial fulfilment for award of Doctor of Agriculture

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(Helena Lina Susilawati)

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General Abstract

Nowadays, global warming is an important issue because it is one of the gravest threats to crop production and environmental sustainability. The agricultural sector is one of the sectors that contribute to global greenhouse gas (GHG) emissions. Rice (*Oryza sativa* L.) is one of the most important staple food crop for over half of the world's population and the demand is expected to increase due to the human population growth. Since irrigated areas are devoted to produce more rice, wastage of the resource, especially water in the rice field should be minimized. It is known that to produce 1 kg of rice in irrigated systems need approximately 4,000-5,000 litres of water. On the other hand, rice cultivation area is expected to reduce due to the conversion of agricultural land into other functions. There is increased competition for land, water, energy, and other inputs into food production. Consequently, the other way to meet increasing agricultural demands is looking toward the new arable land areas including peatland. Expansion of agricultural land is widely recognized as one of the most significant human alterations to the global environment.

In chapter 2, effect of different water treatments on methane (CH_4) and nitrous oxide (N_2O) production were studied under different paddy soils. Water treatments affected to CH_4 and N_2O production potential. Continuous flooding (CF) treatment in incubation experiment produced more CH_4 than flooded-drained-flooded (FDF) in all soil types. However, FDF treatment could not reduce N_2O production in all soil types. Somehow, reducing water from the soils stimulated N_2O production. Five soil types showed different production potential of CH_4 and N_2O which correlated with soil properties. Soil organic carbon (SOC) and ammonium (NH_4^+) showed positive correlations with CH_4 production while total manganese (Mn), total iron (Fe) and nitrate (NO_3^-) showed positive correlations with N_2O production. On the other hand, total Fe and nitrate (NO_3^-) have negative correlations with CH_4 production.

In chapter 3, effect of different water treatments, such as alternate wetting-drying (AWD), site specific AWD (S-AWD) and CF, on CH_4 and N_2O emission were also studied in field experiment. Additionally, yield and water productivity from paddy field was also compared. The AWD and S-AWD are promising methods in irrigated rice cultivation with benefits on water saving, maintaining the productivity and reducing GHG emission comparable to CF irrigation. There were positive correlations between potential production of CH_4 – N_2O and measured CH_4 – N_2O emission (*x*) from Pati's rice soil. It is important that comparative studies should be conducted in different environments to verify this practice as a way to conserve water under conditions of water scarcity while maintaining or increasing crop yields.

In chapter 4, effect of water table and soil amelioration on GHG emissions from peat soil were conducted in columns. Water depths changed the flux of carbon dioxide (CO_2) somehow linearly, while it did not linearly change in CH_4 and N_2O fluxes. There was a positive relationship between water depth and CO_2 emission. Less CO_2 was emitted lower when the water depth near the soil surface. Conversely, a deeper water depth resulted in a slight decrease in the CH_4 emissions. However, the highest N_2O emissions were found at intermediate water depths. The biochar+compost and steel slag+compost treatments increased the CO_2 and N_2O emissions from the peat soil columns. Long-term experiments should be developed to monitor changes that occur over time in response to amelioration at various water depths.

From the above research, it could be concluded that reducing water from rice field could save the water, maintain even increase yield and reduce CH_4 emission. However, it need to be cautious while recommending a particular irrigation regime for rice cultivation in order to avoid substantial emission of one or the other greenhouse gas. Although in this study, reducing water from field could reduce GHG emission. Long-term experiments should be developed for monitoring any changes over years in water depths and ameliorations effects, not only with peat soil but also with soil-crop systems in peat soil to determine how GHG emissions from these type of treatments can be reduced. It might be better to apply ameliorants at higher rates to reach a sustainable reduction in GHG emissions but it should consider the applicability to be used by the farmer.

General abstract (Japanese)

インドネシアにおける無機質土壌と泥炭土壌からの

温室効果ガス放出削減に向けた水位管理と資材添加の検討

ヘレナ　リナ　スシラワティ

温室効果ガス放出による地球温暖化と農業への影響が懸念されており、主要な温室効果ガス放出源である水田からのメタンと一酸化二窒素の生成を抑制しつつ、重要な主食である水稲生産を増加できる技術が求められている。また人口増加に対して、水田への灌漑水供給や水田耕作面積は逼迫しつつある。本研究では、インドネシア、中央ジャワの主要水田土壌中の温室効果ガス生成への水分管理の影響を培養試験で明らかにし、またカリマンタンの泥炭土壌を用いたカラム試験で水位や資材添加がガス生成に及ぼす影響を解明し、さらに実際の水田で水管理のガス放出と水稲生育への影響を総合的に評価した。

第 2 章では、異なる水処理が 5 種類の水田土壌中のメタンと一酸化二窒素の生成能に及ぼす影響を培養実験で調べ、常時湛水が間断湛水に比べ多くのメタンを生成させたが、一酸化二窒素の生成は抑制できなかった。水分を低下させると一酸化二窒素の生成が促進された。5 種類の土壌タイプの違いがメタンと一酸化二窒素の生成能の違いに影響しており、土壌の有機物含量や無機態窒素含量、鉄やマンガン含量と対応していた。

第 3 章では、圃場試験で水管理、すなわち常時湛水と間断灌漑(AWD)、地域特異的間断灌漑(Site specific AWD)の違いがメタンと一酸化二窒素の放出並びに水稲収量と水利用効率に及ぼす影響を検証した。間断灌漑と地域特異的間断灌漑は節水、水稲生産性、温室効果ガス削減の点で常時湛水より優れていた。Pati 水田におけるメタンと一酸化二窒素の生成能と実測された両ガス放出量との間に正の相関関係が認められ、こうした節水管理が他の土壌タイプにおいても水稲収量の維持向上に貢献できることを示している。

第 4 章では、泥炭土壌の水位と土壌改良資材が温室効果ガス放出に及ぼす影響をカラム試験で検証した。地下水位と温室効果ガス放出量との間には相関関係があり、地下水位が高く地表付近の場合、CO2 放出量は少なかった。対照的に地下水位が低いとメタン放出量が微減した。一方、一酸化二窒素放出量は中間的水位で最高に達した。生物炭＋コンポスト，製鋼スラグ＋コンポストの添加は泥炭土壌からの CO_2 と N_2O 放出を増加させた。作物生産性を考慮し、長期的な水位と資材施用の影響を更に研究されるべきである。

以上の研究より、水田での節水管理はメタン放出量の削減と水稲生産性の維持向上に繋がる可能性があるが、メタン以外の温室効果ガスの増加を防ぐような精密な水管理が求められる。またどの添加資材や添加量が泥炭土壌での温室効果ガス削減に有効かを、農民が利用可能かも含めて、長期試験で検討する必要がある。

CHAPTER 1

INTRODUCTION AND OBJECTIVES

1.1 General introduction

Nowadays, greenhouse gas (GHG) emissions and their impact on climate are the important issues for agriculture's sustainability. World agriculture estimated to emit approximately 5.1–6.1 Peta-gram (Pg) CO_2 eq year^{-1}, contributing 10–12% to the total global anthropogenic GHG emissions in 2005 (IPCC 2007a). Agriculture is a source for three primary GHG emissions: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) which contribute in global emissions approximately 60%, 15% and 5%, respectively (Watson *et al.* 1996). These gases long-live in the atmosphere and are the major contributors to positive increases in radioactive forces (IPCC 1996). Each of three trace gases have different global warming potentials (GWP) on a mass basis, which are 298 times higher for N_2O and 25 times higher for CH_4 than CO_2 on a 100-year time scale (IPCC 2007b). Rice fields are important contributors of CH_4 and N_2O emissions. Recently, emission of CH_4 from paddy fields was estimated for about 5-19% of the total CH_4 emissions, while fertilized agricultural soils was estimated for about 13-24% of annual global N_2O emission (Mosier *et al*. 1998; Olivier *et al*. 1998; Kroeze *et al*. 1999; IPCC 2007a). Emission of CO_2 from agricultural sector are mainly due to land uses change (Verge´ *et al.* 2006).

Rice (*Oryza sativa* L.) is one of the most important staple food crop for over half of the world's population (FAO 2013). The demand for rice production is expected to increase due to the human population growth. The global human population is currently growing, and estimates show that it will double by the middle of the next century. Over 90 percent of the world's rice is produced and consumed in the Asian region by 6 countries (China, India, Indonesia, Bangladesh, Vietnam and Japan) comprising 80% of the world's production and consumption (Maclean *et al.* 2002). Indonesia is an agricultural country that had a rice-harvest area of approximately 13.8 million ha in 2014 (Statistics Indonesia 2014). In 1999, Indonesia was the third-largest rice producing country in the world and contributed at approximately 8% of total of world rice production (Coats 2003). One of the responses to increase food production from existing farmland is by intensification of cultivation. These production levels will affect the global environment because rice fields are a source of GHG emissions. If the intensification of rice cultivation is undertaken by using current practices and technologies to increase rice yields, then the GHG emissions from paddy fields will increase substantially if the cultivation technologies are employed without regard for the environment. Many cultivation practices have been improved in order to enhance yield potential and to decrease environmental burdens of paddy rice production. Using high-yielding crop varieties, fertilization, irrigation, and pesticides have contributed substantially to the tremendous increases in food production over the past 50 years (Matson *et al.* 1997).

Rice can be grown under irrigated or rainfed conditions. More than 75% of the rice supply comes from irrigated lowlands rice field (Tuong and Bouman 2003). For producing rice, a continuous flooding (CF) is used for the rice irrigation under the traditional system. In this technique, the paddy fields are inundated all the time starting from transplanting until nearly harvesting. Nowadays, the CF irrigation is getting difficult to be applied due to the decreasing water availability. According to Gleick (1993), the availability of water resources per capita in 2025 are expected to decline by 15–54% compared to 1990 in many Asian countries. There is a major challenge for rice cultivation to grow rice with less water and keep maintain or even increase the yield.

Since irrigated areas are devoted to produce more rice, wastage of the resource especially water in the rice field should be minimized (Oliver *et al.* 2008). One method to save water in irrigated rice field is allowed to dry intermittently of the rice fields instead of keeping them continuously flooded but still adequately supplied with water for the roots of rice plant to maintain rice yields. That method is namely alternate wetting and drying (AWD). The principle of AWD is applied to flood the field in a certain number of days after the disappearance of ponded water. AWD combines the benefit aspect of aerobic and anaerobic rice cultivation. However, the reduced yields and pest control problems (primarily nematodes and weeds) associated with aerobic cultivation is one of the challenges to be addressed in AWD application (Kreye *et al.* 2009).

Recently, the area of rice cultivation is expected to reduce due to the conversion of agricultural land into residential, industrial area and other functions. There is increased competition for land, water, energy, and other inputs into food production. Consequently, the other way to meet increasing agricultural demands is looking toward the areas of arable land. Peatlands cover approximately 3.3% of the land surface area (Hadi *et al.* 2001). Tropical peatlands have been estimated to store up to 15-19% of the global peat carbon pool (Page *et al.* 2011). Approximately 14.9 million ha of peatlands are found in Indonesia and it was estimated 47% of the total tropical peatland areas (Ritung *et al*. 2011; Page *et al.* 2011). Large areas of tropical forest peatland in Indonesia have been converted to agricultural and non-agricultural sectors. Both natural and converted tropical peat soils are sources of CO_2, CH_4 and N_2O due to high levels of Carbon (C)-Nitrogen (N) content and hydrological conditions (Inubushi *et al.* 2003; Arai *et al.* 2014). To be used for agricultural activities, peat soils need to be drained as well as limed and fertilized due to excess water, low macro and micro-nutrient content, high organic acid content and high acidity (Lobb 1997). Expansion of agricultural land is widely recognized as one of the most significant human alterations to the global environment.

1.2 Objectives

1.2.1. General objective

This study was conducted with the general objective to identify the potential option of GHG mitigation in peat soil and mineral soil.

1.2.2. Specific objectives

The specific objectives of this study were:

1. To investigate CH_4 and N_2O production of five paddy soils in different water treatments under laboratory conditions.
2. To investigate the effects of alternate wetting and drying (AWD), site specific AWD (S-AWD) and continuous flooded (CF) on CH_4 and N_2O emission, yield and water productivity from paddy field in Indonesia.
3. To determine the effect of different water depths and soil ameliorants on GHG emissions in peat soil columns.

1.3 Literature review

1.3.1 Global warming and greenhouse gas

Nowadays, global warming is an important issue because global warming is one of the gravest threats to crop production and environmental sustainability. Global warming is one of the most prominent challenges in the present era. Global warming is caused by the increased concentration of greenhouse gas (GHG) in the atmosphere and leads to a phenomenon widely known as greenhouse effect. Greenhouse gas are those that absorb infrared radiation in the atmosphere, trapping heat and warming the surface of the Earth. The three GHG associated with agriculture are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Atmospheric CO_2, CH_4 and N_2O had been recognized as the most important long-live GHG that had significantly contributed to global warming potential (GWP) due to their great radiative forcing. Calculating their GWP therefore depends on the timeframe considered. For a 100-year timeframe, unit masses of CH_4 and N_2O are considered to have 25 and 298 times the GWP, respectively, as a unit of CO_2 (IPCC 2007b).

1.3.1.1. Indonesian national GHG inventory

The National Greenhouse Gases Inventory of Indonesia (NGHGI) reported the calculations of GHG emissions in the second national communication (SNC) on the three main GHG emissions: CO_2, CH_4 and N_2O. There are six emissions categories defined by the Intergovernmental Panel on Climate Change (IPCC): energy, industrial processes, agriculture, land use and land use change and forestry (LULUCF), waste and peat fire-related emissions. In 2000, total Indonesia GHG emissions from six emissions without LULUCF (LUCF and peat fires) reached 556,499 Giga-gram (Gg) CO_2 eq. However, total Indonesia GHG emissions included LULUCF, increased significantly to about 1,205,753 Gg CO_2 eq (SNC 2010). The contribution of each gases on GHG emissions (in CO_2 eq) i.e., CO_2 emitted 940,879 Gg, representing 78% of the total; CH_4 emitted 236,388 Gg or 20% of the total; and N_2O emitted 28,341 Gg or 2% of the total. The main contributing sectors were land use change and forestry (48%), followed by energy, peat fire-related emissions, waste, agriculture and industry approximately around 20, 13, 11, 5 and 3%, respectively (Figure 1.1).

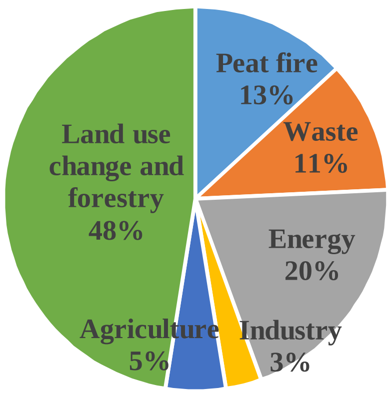


Figure 1.1. National emissions contributions by sector in 2000 (SNC 2010)

1.3.1.2. GHG emissions from agricultural sector

Agriculture is a source for three primary GHG emission: CO_2, CH_4, and N_2O but it can also be a sink for CO_2 through C sequestration into biomass products and soil organic matter. The three main gases are influenced by land management and that are responsible for the potential greenhouse effect. Carbon dioxide, CH_4, and N_2O contribute at approximately 60%, 15% and 5%, respectively, in global emissions (Watson *et al.* 1996).

According to presidential regulation of Republic Indonesia no 61 year 2011, there are 2 targets of agricultural sector to reduce GHG emission in Indonesia until 2030, i.e., reduction of emission target was approximately 0.008 Giga tone CO_2 eq (26%) below business-as-usual (BAU) and approximately around 0.011 Giga tone CO_2 eq for conditional 41% reduction with sufficient international support. There are 3 strategies to achieve the targets of reduction emission, i.e., optimize land and water resources, apply land management and agricultural farming strategies that have lowest GHG emissions and absorb CO_2 optimally and stabilize the water level elevation and arrange foe uninterrupted circulation of water in irrigation network (presidential regulation no 61 year 2011).

a. Carbon dioxide emission

Carbon dioxide emissions from agricultural soils are mainly due to land use change, e.g., when forests are cleared for agricultural development. Soil cultivation and growing annual crops often accelerate the conversion of soil C to CO_2 by soil microbes. After the soils have been cultivated for a few decades, the loss of soil C usually slows down or ceases completely, and the level of soil C becomes stable again, but at a lower percentage (Hutchinson *et al.* 2007). Carbon dioxide, in comparison to CH_4 and N_2O, is cycled in the largest amounts through agricultural cropping systems. Plants consume large amounts of CO_2 through photosynthesis to make food, feed, fibre, and fuel, but all these plant products eventually convert back to CO_2

when consumed or when they decomposed. In agricultural sector, the net emission of CO_2 is small in comparison to its total cycling and is mostly due to energy use on-farm or in the manufacture and transport of agricultural products.

b. Methane emission

Methane is a reactive and radioactively trace gas that has strong infrared absorption band characteristics, which is contributing to changes in atmospheric chemistry and may cause global warming (Bouwman 1991). The main contributors of CH_4 production from agricultural sector are rice paddies, ruminants, landfills, natural wetlands and sediments (Yang and Chang 1998). Figure 1.2 shows the mechanism of CH_4 production, consumption and transport from rhizosphere to atmosphere. In paddy fields, CH_4 emission is end product of the organic matter degradation and the result of complex interactions between rice plants and soil microbes under anaerobic conditions (Cicerone and Oremland 1988; Neue and Sass 1994; Conrad 1996; Conrad 2007). Anaerobic condition emit 80% of atmospheric CH_4 by methanogenic bacteria during digestion of organic matter in submerged soils (Ehhalt and Schmidt 1978). The rice plant provides methanogenic substrate through root exudates, decaying root tissues and to a lesser extent, litter fall from above ground parts and creates an active CH_4-oxidizing site in the rhizosphere (Wassmann and Aulakh 2000). Root exudates of the rice plant consists of carbohydrates, organic acids, amino acids and phenolic compounds (Aulakh *et al.* 2001). Organic matter fermentation result acetate ($CH_3CO_2^-$), CO_2, hydrogen (H_2), propionate as well as other fatty acids which are the main substrates to produce CH_4 by methanogenic bacteria (Kruger *et al.* 2002; Conrad *et al.* 2010). Acetate is one of the main precursors of CH_4 production in rice soils through demethylation (Sigren *et al.* 1997) and can be derived either from root exudation (Lin and You 1989) or from fermentation (Neue and Roger 1993). Moreover, it provides methanogenic bacteria with plenty of precursors to produce CH_4 and accelerates the decline of soil redox potential (Eh), thus forming a favourable environment condition for growth of methanogens, which in turn promotes formation of CH_4 and further affects CH_4 oxidation capacity in the field (Bender and Conrad, 1995; Arif *et al.* 1996). The requirements for soil reduction are the absence of oxygen, the presence of decomposable organic matter, and anaerobic bacterial activity. In the absence of oxygen, facultative and obligate anaerobes use nitrate (NO_3^-), manganese (Mn(IV)), iron (Fe(III)), sulphate (SO_4^{2-}) dissimilation products of organic matter, CO_2, nitrogen (N_2), and even hydrogen (H^+) ions as electron acceptors in their respiration reducing NO_3^{2-} to N_2, Mn(IV) to Mn(II), Fe(III) to Fe(II), SO_4^{2-} to hydrogen sulphide (H_2S), CO_2 to CH_4, N_2 to NH_4^+, and H^+ to H_2 (Ponnamperuma 1972).

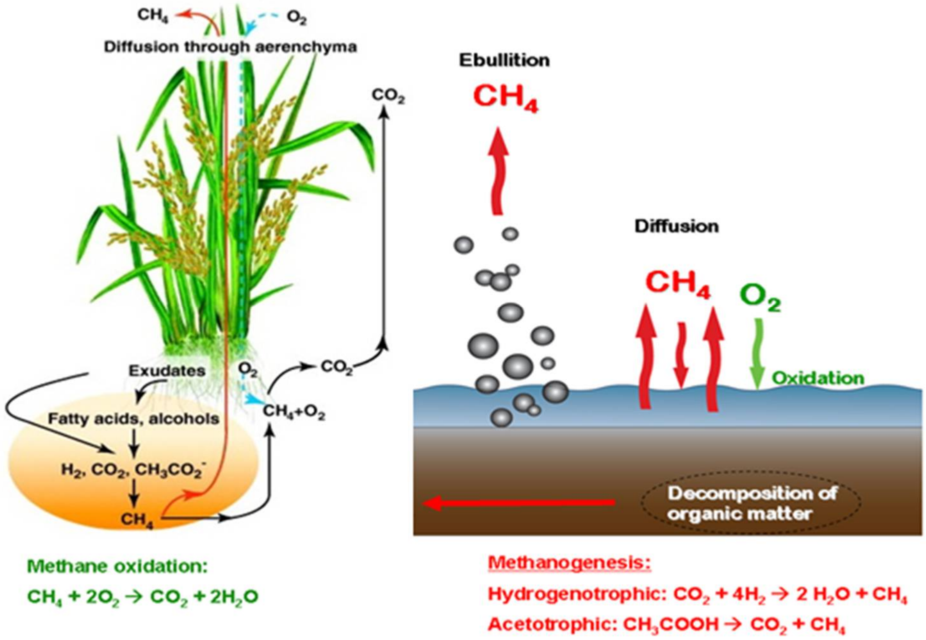


Figure 1.2. Methane production, consumption and transport in rice field (Philippot and Hallin 2011; https://www1.ethz.ch/ibp/research/environmentalmicrobiology/research/Wetlands)

Under anaerobic condition, more than 90% transport CH_4 from rhizosphere into the atmosphere and the oxygen diffusion into roots is mediated by the aerenchyma and intercellular space system of rice plants in leaf blades, leaf sheaths, culm and roots (Raimbauit *et al.* 1977; De Bont *et al.* 1978; Inubushi *et al*. 1989; Schutz *et al.* 1989) and the rest is released by the bubbles. Methane emission from rice field varies substantially among water management strategies. Globally, irrigated rice accounts for 70-80%, rainfed for 15% and deepwater rice for about 10% of the CH_4 produced from rice (Wassmann *et al.* 2000). Upland rice is not considered a significant source of CH_4 due to less water in the field (Neue 1997). Water management during the production of rice is a key factor in minimizing CH_4 during rice production. Draining the water and allowing the soil to become aerobic allows oxidation of CH_4 and reduces CH_4 production (US EPA 2006).

c. Nitrous oxide emission

Nitrous oxide is important trace gas in the global N cycle. Increases in the atmospheric concentration of N_2O contribute to global warming as well as directly to the destruction of the stratospheric ozone layer (IPCC 1996). Nitrous oxide emissions from paddy fields represent a substantial source of atmospheric N_2O, although in small quantity compared with those from upland systems (Xing 1998). According to Yan *et al.* (2000), rice plants are an important pathway of N_2O emission in the presence of flood-water. In the presence of floodwater, N_2O emission is released around 87% through the rice plants. While in the absence of floodwater, N_2O is emitted through the soil surface, with only 17.5% on average released through the plants.

The production of N_2O in wetlands was shown in Figure 1.3. Nitrous oxide is by-products of microbial nitrification or intermediate products of denitrification processes (Mosier and Kroeze 2000). The N_2O emission from soils by nitrification and denitrification processes depends on environmental and agricultural management factors, such as rain, temperature, fertilization, irrigation, and heavy metal accumulation, as well as on soil properties such as pH, organic matter content, and particle size (Khalil *et al.* 2003). Nitrification is commonly defined as the biological oxidation of NH_4^+ to NO_3^- with nitrite (NO_2^-) as an intermediate (Bremner 1997). Ammonium produced by mineralization of soil organic matter (SOM) which is utilized by soil microorganisms and plants, this condition make the NH_4^+ concentration in agricultural soils is generally low. However, the application of urea and NH_4^+ make nitrification becomes the most active process in soils and quickly emit N_2O after N fertilizer is applied in the soil (Nishio and Fujimoto 1990; Cheng *et al*. 2002). The availability of oxygen (O_2) in soil is one of the main factors regulating nitrification, denitrification and the release of N_2O. Denitrification occurs when NO_3^- is present in anaerobic microsites developed wherever microbial demand for O_2 exceeds diffusion-mediated supply (Arah and Smith 1989). Denitrification in soils also consumes N_2O through the reduction of N_2O to N_2. Hence, this bacterial process may serve either as a source or as a sink of N_2O. Nitrification also contributes to N_2O emission due to fertilizer addition to soils during the oxidation of NH_4^+ or hydroxylamine (NH_2OH) to NO_3^- (Pathak 1999).

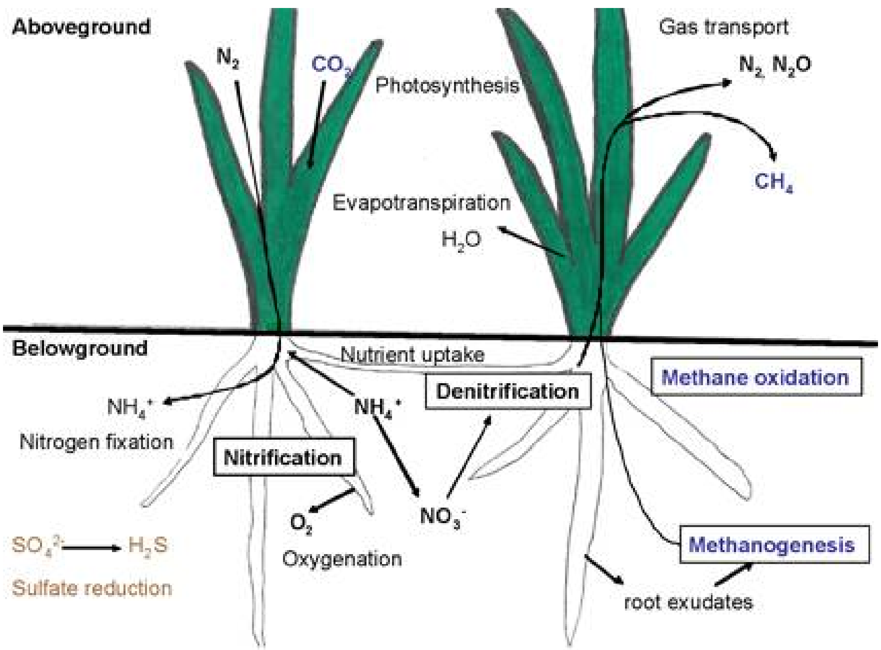


Figure 1.3. N_2O emission from wetland, http://geology.usgs.gov/postdoc/profiles/moseman/

1.3.2. Rice

Rice (*Oryza sativa (L.*)) is cereal food for feeding of more than half of the world's population, the most important food crop in many developing countries, and has also become a major crop in many developed countries where its consumption is predicted to increase by about 24% in the next 20 years (Van Nguyen and Ferrero 2006; Patel *et al.* 2010). In 2008, the rice consumption in Asia reached around 90% of the world rice consumption (IRRI 2009). Agricultural sector has a strategic position in Indonesia's economy. In 1999, Indonesia was the third-largest rice producing country in the world and contributed at approximately 8% of total of world rice production (Coats 2003). Although as third largest rice producing country, Indonesian rice production still not enough to fulfill its consumption. Figure 1.4 shows about Indonesian rice production, total consumption and harvested area, 1960-2015 (Ito 2015). Since 1960, the Indonesian consumption of rice exceeded the rice production likely occurred because Indonesia has very high human population and it increase very fast as well as limited area to change into rice field and dietary change.

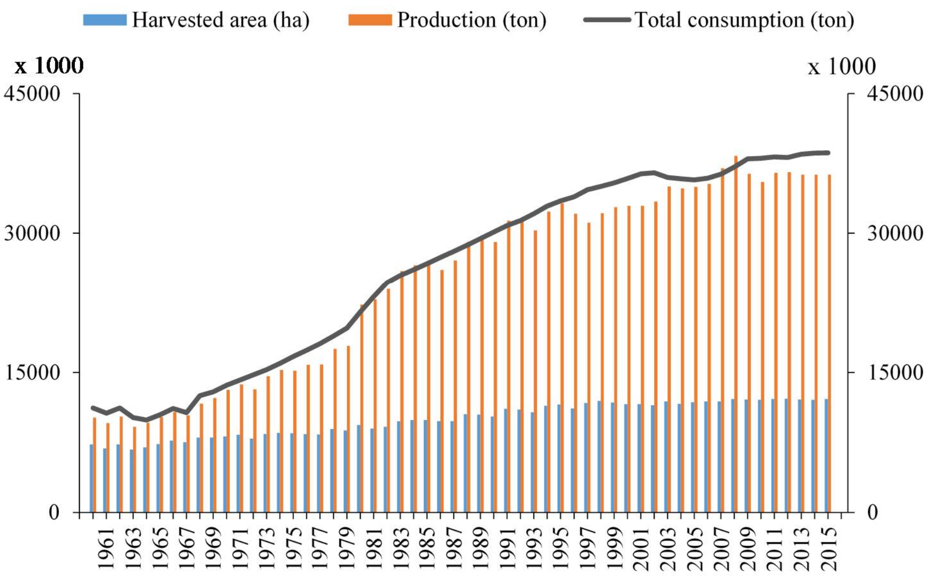


Figure 1.4. Indonesian rice production, total consumption and harvested area, 1960-2015 (Ito 2015)

Rice is produced in a wide range of locations and under a variety of climatic conditions. The rice plant's environments have been divided into agro-ecological zones (AEZs): tropical regions, subtropical regions with summer or winter rainfall, and temperate regions (Maclean *et al.* 2002). Rice ecosystems can be defined irrigated, rainfed, lowland, upland, and flood-prone (Dobermann *et al.* 2004). Rice ecosystem in Indonesia was 54% irrigated rice area, 35% rainfed and 11% upland (Maclean *et al.* 2002). Irrigated rice production is the leading consumer of water in the agricultural sector, and rice is the world's most widely staple crop consumption, therefore finding ways to reduce the need for water to grow irrigated rice should benefit both producers and consumers. Most rice varieties maintain better growth and produce higher grain yields when grown in a flooded soil than when grown in dry soil (De Datta 1981). Recently, the concern about the sustainability of food production has increased because of the increasing of world population and the growing of environmental issues.

Rice growth duration is 3–6 months, depending on the variety and the environment under which it is grown. During this time, rice completes two distinct growth phases: vegetative and reproductive. The average rice yields in rice-growing countries range from less than 1 to more than 6 t/ha. There are many factors that influence the rice yield. Temperature, solar radiation, and rainfall influence rice yield by directly affecting the physiological processes involved in grain production, and indirectly through diseases and insects (Yoshida 1981). There are 16 essential elements and divided into major and minor elements. The major elements, carbon (C), hydrogen (H), oxygen (O), nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), magnesium (Mg), and sulphur (S), are needed by plants in relatively higher amounts than the minor elements, Fe, Mn, copper (Cu), zinc (Zn), molybdenum (Mo), boron (B), and chlorine (Cl) (De Datta 1981). The nutrients can be made available to plant roots by contact exchange and soil solution. Based on rice disease-causing agents, the diseases of rice are classified into four groups i.e., fungus, bacteria, virus, and nematode (Ou 1979).

1.3.2.1. Water scarcity

Water is the primary resources for agriculture and food production. Water is a critical and the most important factor in rice production. Without water, no crops can be grown. About 75% of rice areas in the world are under continuous flooding (Van der Hoek *et al.* 2001). Rice grown under traditional practices in the Asian tropics and subtropics requires between 700-1500 mm of water per cropping season depending on soil texture (Bhuiyan 1992). It is known that in irrigated systems, approximately around 4,000-5,000 litres of water are used to produce 1 kg of rice in many areas (Tabbal *et al.* 1992). Presently, irrigation water efficiency is generally low. Efficiency for the flood irrigation practiced in paddy fields can be as low as 20% (Abdullah 2004). The competition related to the demand for water between agriculture and other sectors such as industry, environment, has become acute. Scarcity of water for agricultural production is becoming a major problem in many countries, particularly the world's leading rice-producing countries. Rainfall patterns in many areas are becoming more unreliable, with extremes of drought and flooding occurring at unexpected times due to climate change. Attempts to reduce water in rice production may result in yield reduction and may threaten food security in Asia. Reducing water input for rice will change the soil from submergence to greater aeration. These shifts may have profound – and largely unknown – effects on the sustainability of the lowland rice ecosystem.

1.3.2.2. Water management

Doorenbos and Kassam (1986) stated that water content below 70% of saturation cause yield decreases and at 50% could give a 50-70% yield reduction. At 30% of saturated water contents, zero yield is obtained, while at 20%, the rice plants will wilt and die. The challenge in paddy rice ecosystem to address problems of water scarcity, researchers had been looking for ways to develop the approaches that allow rice production to be maintained or increased in the face of declining water availability. One such strategy to address this problem in rice field is the use of alternate wetting and drying (AWD) irrigation. Alternate wetting and drying is a water-saving technology in irrigated fields that applied alternately flooded and non-flooded condition in the field. The number of days of non-flooded soil in AWD between irrigations can vary from 1 day to more than 10 days, irrigation is applied after soil water potential has reached –10 to –30 kPa at 15 cm depth or shallow groundwater tables have gone to 10 to 40 cm depth, depending on growth stages of rice (Zhang *et al.* 2009). AWD can lower water use for irrigated rice by 35%, increase rice yield by 10% relative to continuous flooding (Yang *et al.* 2009; Zhang *et al.* 2009). However, other studies reported that AWD resulted slightly lowers yield (Sudhir-Yadav *et al.* 2012; Yao *et al.* 2012). One of the challenges on AWD adoption by the farmer is farmers do not always receive water at the time of demand, since irrigation is executed by a pump operator or other people who in charge on water management in that site (Kürschner *et al.* 2010).

1.3.2.3. Soil amendment/ ameliorant

Soil amendment is the process of modifying soils to provide what the native or existing soils do not naturally provide. The amendment required can vary depending upon the existing soil and the traits of the soil that require alteration, for examples improving the drainage of a heavy clay soil, increasing the nutrient holding capacity of a highly sandy soil or repelling the negative effects of a saline soil near the coast with the application of calcium. Soil amendment mostly are conducted to enhance the nutrient status of the soil and to improve crop yield as well as to reduce GHG emission, e.g., biochar, manure, steel slag fertilizer (Table 1.1).

Steel slag, a by-product of the steel industry, contains of high iron, silica and calcium. Steel slag could enhance canopy photosynthesis, increased biotic and abiotic stress resistance, and contributions to healthy growth and high yields because it contain a high silica content (Ma and Takahashi 2002). Some studies have shown that the application of steel slag fertilizer with high silica contents reduced CH_4 emissions because of aerenchyma enlargement (Furukawa and Inubushi 2004; Jackel *et al*. 2005; Ali *et al*. 2008). Steel slag could be used as an oxidizing agent to suppress CH_4 emissions from rice fields. Electron acceptors such as NO_3^-, Mn_4^+, Fe_3^+ and SO_4^{2-} can decrease CH_4 production because of inhibitory and competitive effects with different microorganisms for common electron donors (Jakobsen *et al*. 1981; Achtnich *et al*. 1995). According to Ali *et al*. (2014), the application of silicate fertilizer with organic amendments decreased CH_4 flux, increased grain yield and improved soil quality. However, the application of steel slag as a soil ameliorant results in a trade-off between N_2O and CH_4 emitted from paddy fields. Some studies have shown that applications of steel slag reduced CH_4 emissions (Furukawa and Inubushi 2004; Ali *et al*. 2008) and stimulated N_2O emissions (Huang *et al.* 2009; Singla and Inubushi 2013). Based on Huang *et al.* (2009), using the same product could stimulate N_2O emissions by enhancing the nitrification rate. Conversely, steel slag applications suppressed N_2O emissions from rice fields (Susilawati *et al*. 2015).

Biochar application to soil is believed to improve soil fertility as well as sequester C to mitigate climate change (Lehmann *et al*. 2011). Biochar applications significantly decrease N_2O emission and increase CH_4 emission from paddy fields in mineral soil (Zhang *et al.* 2012; Singla and Inubushi 2014). Existing reviews show conflicting results with respect to GHG emission following biochar application and the mechanisms remain debatable (Lehmann and Sohi 2008; Wardle *et al.* 2008; Mukherjee and Lal 2013). Another suggestion for improving soil fertility is the application of organic matter (Glaser *et al*. 2002). Manure apparently has a strong ability to mitigate soil acidity and enhance Ca uptake in a tropical forage legume (Hue 1992). However, organic matter can be mineralized very rapidly under tropical conditions (Tiessen *et al.* 1994).

Table 1.1. The emission of CO_2, CH_4 and N_2O from soil ameliorants in different land use in field

Type of ameliorants	Application rate (Mg ha^{-1})	Indicator plant	Country	Soil types	Range of			References
					CO_2	CH_4	N_2O	
					mg C m^{-2} hour^{-1}		µg N m^{-2} hour^{-1}	
Compost	12	Paddy	Japan	Gley soil		5.07		Yagi and Minami 1990
				Andosols		2.53		
Rice straw	6	Paddy		Gley soil		12.80		
				Andosols		4.27		
				Peat soil		21.73		
Biochar (wheat straw)	10-40	Paddy	China	Entic Halpudept	54.4-65.9	1.02-1.82	83.33-129.17	Zhang et al. 2012
Green manure	20	Paddy	India	Typic Ustochrept		2.99		Khosa et al. 2010
Rice straw compost	10					0.80		
Wheat straw	10					7.08		
Farmyard manure	20					3.41		
Wheat straw	3.75-4.8	Paddy	China	Typic Epiaquepts		4.77-20.54	1.54-8.92	Ma et al. 2009
Steel slag	2-8	Paddy	China	No information		1.59-2.29	12.14-28.54	Wang et al. 2015
Urea, rice straw compost	0.22; 2, respectively	Paddy	Bangladesh	Clay loam		3.90		Ali et al. 2014
Urea, rice straw compost, silicate fertilizer	0.17; 2; 0.3, respectively					3.71		
Urea, sesbania, silicate slag	0.17; 2; 0.3, respectively					3.79		
Urea, azolla anabaena, silicate slag	0.17; 2; 0.3, respectively					3.55		
Urea, cattle manure compost, silicate slag	0.17; 2; 0.3, respectively					3.67		
Manure	4	Paddy	Indonesia	Peat soil	162.5	9.21		ICCTF 2011
Steel slag fertilizer (Pugam A)	0.75				163.6	9.39		
Steel slag fertilizer (Pugam T)	0.75				218.2	8.52		
Mineral soil	4				179.5	11.66		
Biochar	20-40	Maize	China	Inceptisols	55.4-62.2	-0.03 to -1.88	22.00-47.73	Zhang et al. 2015
Coated fertilizer	0.046	Oil palm	Indonesia	Peat soil	153		272	Sakata et al. 2015

1.3.3. Peat soil

1.3.3.1. Peat soil characteristic

Peat soils cover more than 420 million ha worldwide (Clymo 1987) which is around 3% of the earth's surface and contain around 10% of the total C stored in peatlands (Immirzi *et al.* 1992; Maltby and Proctor 1996). Around 36 million ha of it can be found in the tropics and subtropics (Andriesse 1988). Peat consists of dead, partially decomposed plant remains that that has accumulated over thousands of years in waterlogged environments that lack oxygen (Wösten *et al.* 2008). The plant material from which the peat is derived does have some influence on the chemical composition of the peat. Aboveground plant production is believed to be the primary source of peat (Clymo 1983). Tropical peatlands are predominantly forested with no moss cover (Rieley and Ahmad-Shah 1996). The high temperatures and more aerobic conditions on the surface of tropical peatlands may speed up the decomposition of leaves and wood, suggesting that roots could be more important to peat accumulation in the tropics. The accumulation rates range of tropical peatland approximately 4–5 mm $year^{-1}$ until 5–10 mm $year^{-1}$, significantly faster than in most temperate and boreal peatlands because tropical peatlands occur in consistently hot and often humid conditions. However the rate of accumulation temperate and boreal peatlands approximately around 0.5–1 mm $year^{-1}$ (Gorham 1991, Maas 1996 and Gorham *et al.* 2003).

Important physical properties of the peat soils are bulk density, porosity, water holding capacity, subsidence, and irreversible shrinkage. Normally, higher degree of peat maturity (fibrists < hemists < saprists) will be followed by the high degree of bulk density (BD) and C organic content. Intensive management on peat land and environmental drainage could significantly influenced on BD and C density of peat. According to Wahyunto *et al.* (2010), the types of mineral materials substratum − non-sulfidic clay, sulfidic (marine) clay or sandy substratum− underlying the peat determine the peat fertility. The non-sulfidic clay are commonly found in inland peats, and considered as the better substratum. However, the quartz sand substratum determine low to very low peat soil fertility. Moreover, the substratum peat soil in shallow peats, sulfidic and exposed to aerobic condition, there is a possibility of acid sulphate soils formation and a very poor peat for crop cultivation (Wahyunto *et al.* 2010). Peat is source of organic materials and particular it contains sizeable quantities of lignin, bitumens and humic acids (Delicato 1996). Tropical peatlands are widely distributed under waterlogged and acidified conditions. Tropical peatlands have a high porosity and, as a consequence, a high water-holding capacity that provides them with an important water regulation function with respect to downstream tropical lowlands.

1.3.3.2. Peat soil and greenhouse gas emissions

Peatlands play important roles in the global cycling of C as they are net sinks of atmospheric CO_2 and under natural decomposition release by-product such as N_2O, CH_4 and CO_2 (Gorham 1991). The total C stored in tropical peatland is about 83.3 Gt where 44.5 Gt or about 53.1% is found in Indonesia across the three main islands, i.e., Sumatra, Kalimantan and Papua (West Papua) with total C stored of 18.3 Gt (41.1%), 15.1 Gt (33.8%) and 10.3 Gt (23%), respectively (Saharjo 2011). It has been estimated that peat and forest degradations contribute to about 45% of total GHG emissions from Indonesia (Ridlo 1997). This considerable contribution of peat on total GHG emissions has put peat soil as a target for C emission reduction. Peat management is targeted to reduce 9.5-13% of GHG emissions from Indonesia by year 2020 (Las and Surmaini 2010).

Table 1.2. The emission of CO_2, CH_4 and N_2O from different land use in Indonesia.

Location	Land-use	Range (min-max) of CO_2	CH_4	N_2O	References
		mg C m^{-2} hour^{-1}		µg N m^{-2} hour^{-1}	
South Kalimantan, Kalimantan	Abandoned upland crops field, abandoned paddy fields, secondary forest	113 to 176	0.07 to 0.22	-12.56 to -0.42	Inubushi *et al.* 2003
Jambi, Sumatera	Drained forest, cassava field, upland paddy field, lowland paddy field	30 to 266	0.10 to 4.24	3.77 to 62.21	Furukawa *et al.* 2005
South Kalimantan, Kalimantan	Secondary forest, paddy field, upland crops field, abandoned paddy field, abandoned upland, rice-soybean rotation field	30 to 804	-0.08 to 8.01	-30 to 1040	Hadi *et al.* 2005
Riau, Sumatera (1°30'N, 103°40'E)	Sago palm	24 to 150	-0.04 to 0.99	nd	Watanabe *et al.* 2009
Central Kalimantan, Kalimantan (02°21'S, 114°02E and 2°19'S, 114°01E)	Natural forest, regenerated forest, burned forest, grassland cropland	nd	nd	5 to 2.957	Takakai *et al.* 2010
Central Kalimantan, Kalimantan (2°20'27.74''S, 114°2'16.48''E)	Ferns, sedges, pulp wood	10 to 455	0 to 158.63	-1.05 to 292.29	Jauhiainen and Silvennoinen 2012
South Kalimantan, Kalimantan	Paddy, oil palm, vegetable	-0.38 to 1.30	0.02 to 0.19	-7.78 to 52.34	Hadi *et al.* 2012
Kalampangan, Kalimantan (2°17'-2°21'S, 113°54-114°01'E)	Undrained natural forest, drained forest, burned forest, cropland	80 to 349	-0.02 to 0.36	0.01 to 13.13	Arai *et al.* 2014a, 2014b
Central Kalimantan, Kalimantan (02°12'26''S, 55'00''E)	Flooded forest, drained forest, flooded burnt site, drained burnt site	108 to 340	0.01 to 5.75	-2.40 to 8.10	Adji *et al.* 2014
Tatau, Malaysia (02°57.924N, 112°45.851'E)	Oil palm	90 to 223	nd	131 to 523	Sakata *et al.* 2015
Serawak, Malaysia (2°49'N, 111°51E; 2°47'N, 111°50E and 2°49'N, 111°56'E)	Sago, oil palm, forest	46 to 533	nd	-3.4 to 176.3	Melling *et al.* 2005, 2007

nd: no data

Decomposition of drained peatlands in Indonesia is estimated to cause 632 Tera gram (Tg) year^{-1} CO_2 emissions (range 355–874 Tg year^{-1}), which will likely increase every year for the first decades after 2000 unless peatland use practices are changed (Hooijer *et al.* 2006). The cultivation of tropical peatlands has been considered to be a large source of N_2O emissions and the IPCC (2000) collected the data from boreal and temperate regions and estimated that direct total N_2O emissions from mineralization of soil organic nitrogen in cultivated organic soils approximately 16 kg N ha^{-1} year^{-1}. The CO_2, CH_4 and N_2O emission from different land uses in peat soil in Indonesia has been observed by many studies (Table 1.2)

1.3.3.3.Peatland for agriculture

Peat soil are fragile ecosystems with important biological and hydrological functions. Demand to expand agriculture are likely to lead to further deforestation. This conditions resulted the water adjustment and soil improvement that suitable for agriculture or for other land use. The process of drainage peatland is shown in Figure 1.5. Natural peatlands is a sequester carbon, waterlogged dome and mostly water tables lies near soil surface. Agricultural use of peatlands need to lowering of the water table, increased aeration, and changes in plant. When drainage starts as implication of land use change, it has led to a number of effects including increasing of soil decomposition, c losses, and CO_2 emission. Then, if drainage continued lead to peat subsidence and CO_2 emission, which is sign of carbon loss due to increased organic matter decomposition (Jauhiainen and Silvennoinen *et al.* 2012). It has known that peat soil is irreversible soil, thus lowering water causes peat shrinkage, biological oxidation, loss of carbon stock and high risk of smouldering peat fire. Sometimes, agricultural use of peatlands often destroyed their ecological character and the ecosystem services that go with it. This is the consequences of using pristine peatland as agricultural functions. Figure 1.6

shows that large number of deforested peatland in South-East Asia and still remaining peatlands can be expected to be logged and drained in the next few decades. According to Hooijer *et al.* (2006) that the area used for oil palm plantations, currently estimated at 20 000 km^2, is expected to more than double to a surface area of 50 000 km^2 by 2050.

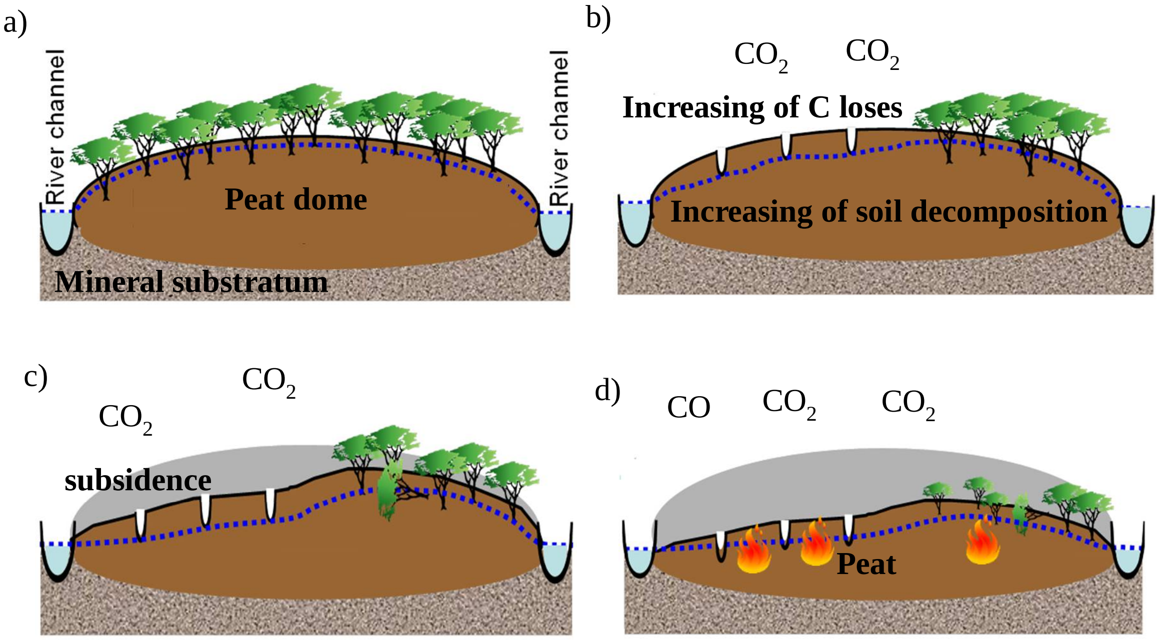


Figure 1.5. The process of drainage tropical peatland, a) natural peatland; b) when drainage starts; c) and d) when drainage continue (source: modify from http://blogs.helsinki.fi/jyjauhia/peat-in-agriculture-and-forestry)

Several management strategies such as soil conservation practices, incorporation of crop residues, use of composts, minimum tillage and others soil-crop managements hold promise for achieving GHG mitigation and adaptation (Hobbs 2007; Delgado *et al.* 2011). Murdiyarso *et al.* (2010) reported that intensifying existing production of oil palm and locating new plantations in degraded secondary forests and grasslands in peat swamp forest in Indonesia can both satisfy demands and reduce greenhouse gas emissions. Thornton and Herrero (2010) observed that extensive tropical livestock systems and conclude that management options could mitigate a maximum of ~7% (417 Mt CO_2 eq or 0.417 Pg CO_2 eq) of the global agricultural mitigation potential to 2030 without reducing production. Management options to enhance food production commonly involve trade-offs among multiple objectives. Synergistic options to meet multiple objectives although less common, when and where they do exist should be done to achieve not only food security and sustainability but also environmental friendly.

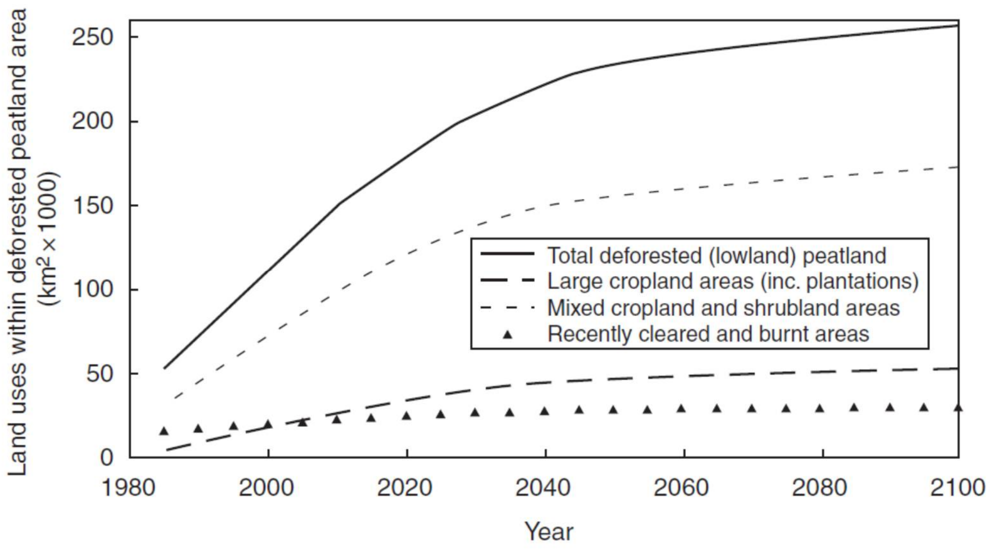


Figure 1.6. Current trends and future projections of land use within deforested peatlands in South-East Asia (Verhoeven and Setter 2009).

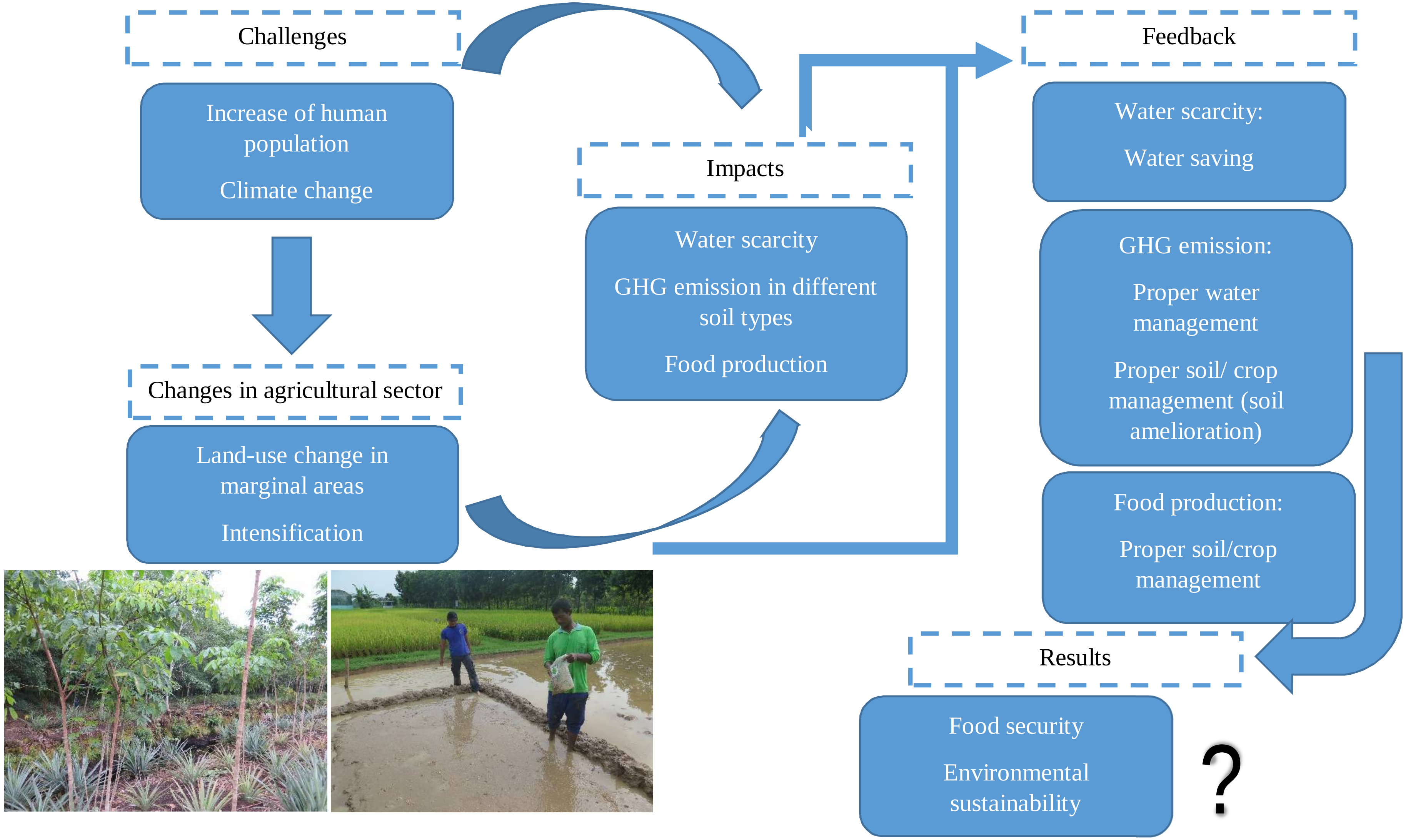


Figure 1.7. The framework study of water managements and soil ameliorations in mineral soil and peat soil

Chapter 2

METHANE AND NITROUS OXIDE PRODUCTION CAPACITIES FROM DIFFERENT RICE SOILS UNDER DIFFERENT WATER TREATMENTS

2.1. Introduction

Methane and N_2O are considered the major sources of GHG, emitted mainly from flooded rice fields because the coexistence of aerobic and anaerobic condition (Reddy *et al.* 1989). Methane production in soils generally occurs under strictly anaerobic condition. On the other hand, N_2O is produced from nitrification under aerobic conditions, and denitrification under moderately anaerobic conditions. Generally, there is a trade-off between CH_4 and N_2O production in rice soils and this condition makes more challenge on reducing the production of one gas but not to increase the production of the other.

The process of wetting and drying of the soil change soil structure therefore likely to affect the biological processes that lead to C and N transformations and the biogenic gases production (Beare *et al.* 2009). Flooding and unflooding field influence root activity, photosynthesis and respiration of rice plants. In a soil profile, CO_2 is produced by respirations of plant root and microorganisms. The CO_2 is partly leached (Minamikawa *et al.* 2010). However, many of the data obtained so far are not sufficiently detailed to examine CO_2 exchange in rice paddies (Liu *et al.* 2013).

There are important factors in soil that control CH_4 and N_2O production from rice fields. Methanogenesis and N_2O production are influenced by physical and biochemical factors in the soil, such as soil pH, redox potential, organic matter content, temperature, and soil moisture content. The content of soil oxidants (O_2, NO_3^-, Mn_4^+, Fe_3^+, SO_4^{2-} and CO_2) used as electron acceptors for organic matter degradation contributes significantly to these processes (Yu *et al.* 2001). The reduction of various oxidants in homogeneous soil suspensions occurs sequentially at corresponding soil redox potential values (Ponnamporuma 1972). A better understanding of this relationship is needed in order to be able to possibly mitigate the emission of these important GHGs through changes in agricultural practices.

The objective of this study was to investigate CH_4 and N_2O production of five paddy soils of Indonesia in different water treatments under laboratory conditions.

2.2. Materials and Methods

2.2.1. Soil samples collection

Soil samples were collected from irrigated rice areas in Central Java, Indonesia (Figure 2.1). The soil samples were taken from surface layer (0 to 15 cm) at all sites. The samples were collected in March 2014. Soil maps with a scale of 1:250,000 from Indonesian Center for Agricultural Land Resources Research and Development (ICALRRD) were used to determine the soil sampling sites. Five sites of paddy fields, (i) Klaten, (ii) Boyolali; (iii) Grobogan; (iv) Demak and (v) Pati district, were classified as typic dystrudepts, typic hapludants, typic epiaquepts, typic epiaquepts and aeric endoaquepts, respectively. The soil samples were brought to Indonesian Agricultural Research Institute (IAERI), Jakenan, Indonesia. Characteristics of soil used are listed in Table 2.1.

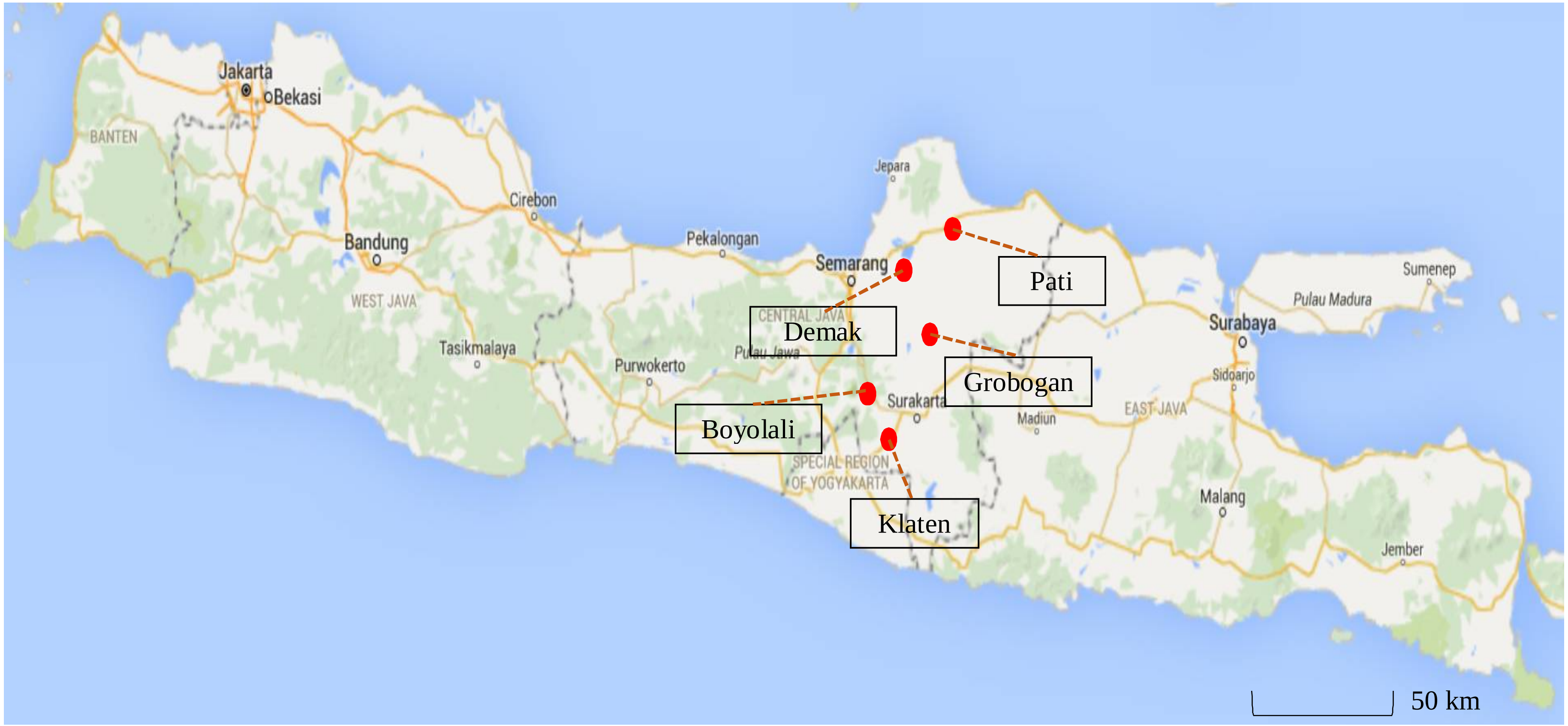


Figure 2.1 Map of Java Island. Red dot colour is the location of soil sampling.

2.2.2. Soil incubation

Soil samples were air-dried, and ground to pass through a 2-mm stainless steel sieve with removal of visible plant residues. Each of the soils was taken in a glass bottle and pre-incubated for 7 days. Pre-incubation of the soil samples were carried out in the dark at 30^0C and in aerobic condition. After pre-incubation finished, 20 grams of soil sample were put into a 120-ml beaker and 40 ml of distilled water was added to keep the soil saturated. Magnetic bar beakers bottles were closed tightly with butyl rubber stoppers and silicone grease sealant. The stoppers had 4 holes fitted with stoppered glass tubing to facilitate flushing of beaker headspace with N_2, collection of gas samples through septa via syringes and monitoring of pH-Eh by inserting suitable electrodes. Headspace was flushed with N_2 at a rate of 300 ml min^{-1} for at least 10 min to stimulate soil reduction. The beakers were kept in incubator and maintained at temperature of 30^0C for 57 days of incubation (Mitra *et al.* 2002). Two sets of bottles were used for gas sampling: one set was used as continuous flooded (CF) and the other was used as flooded-drained-flooded (FDF). All measurements were carried out in triplicate.

During the drying phase, water were removed from the FDF incubation bottles for 7 days (Figure 2.2). Gas samples were collected every day for both treatments. After 7 days, soil incubations were rewetted and gas samples were collected once a week. The NO_3, NH_4, Fe total and Mn total analyses were repeated once a week on the remaining soil incubation at the termination of the experiment.

Table 2.1. Characteristics of the soils used

Parameters	**Klaten**	**Boyolali**	**Grobogan**	**Demak**	**Pati**
Soil order	Typic dystrudepts	Typic hapludants	Typic epiaquepts	Typic epiaquepts	Aeric endoaquepts
Latitute	$07^0$36’23.4”	$07^0$26’39.1”	$07^0$03’13.0”	$06^0$52’10.6”	$06^0$46’42.1”
Longitude	$110^0$42’24.3”	$110^0$41’41.9”	$110^0$46’17.0”	$110^0$42’58.6”	$111^0$11’52.0”
Sand (%)	29	15	11	15	43
Silt (%)	34	19	24	20	40
Clay (%)	37	67	65	65	17
pH (H_2O)	6.4	7.8	6.8	7.4	5.6
CEC (cmol kg^{-1})	11.38	35.66	30.64	20.68	7.93
Active Fe (%)	1.25	0.56	1.05	0.4	0.2
Available P (mg kg^{-1} soil)	27.1	10.0	22.7	28.8	4.8
Total N (g kg^{-1} soil)	1.1	1.1	0.7	0.7	0.2
Total C (g kg^{-1} soil)	10.8	12.5	8.2	2.0	1.6

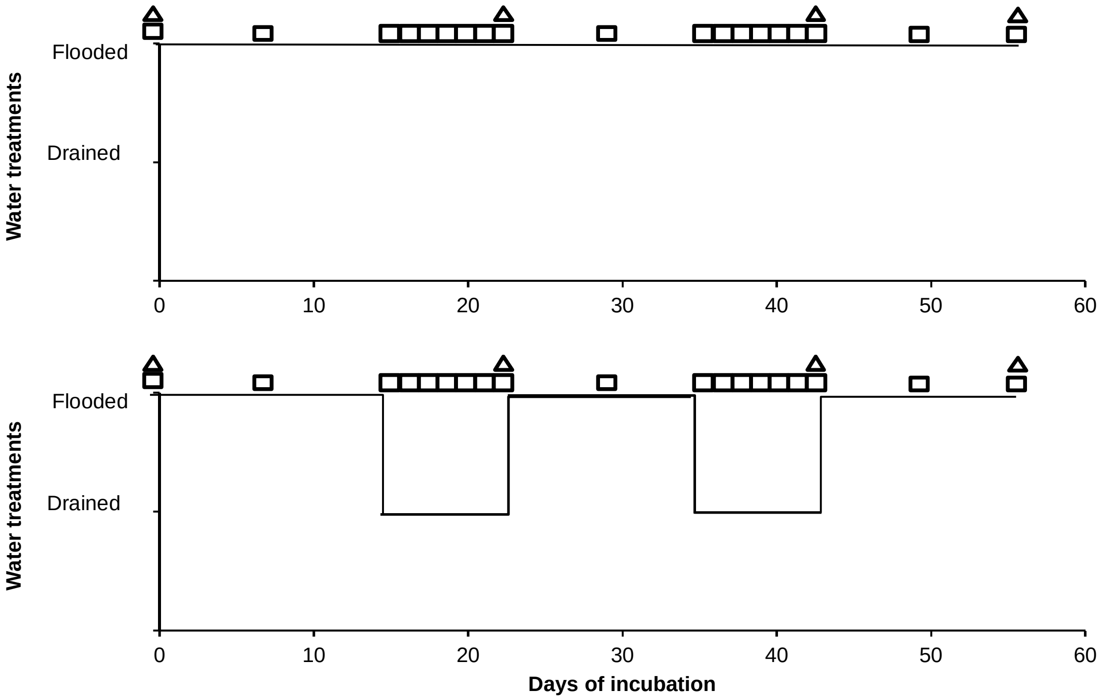


Figure 2.2. Schematic diagram of water treatments during incubation experiment

2.2.3. Gas sampling

Gas samples were taken and analysed periodically at 1, 8, 15, 16, 17, 19, 22, 30 36, 37, 38, 40, 50 and 57 days of incubation (DOI). During time 0 (C_0), the beaker were stirred and flushed with N_2 for 2 minute before gas sampling to flush out accumulated gas from the beaker headspace, then the beaker were closed and the gas samples in the headspace of the beaker were taken using a syringe. After 24 hours (C_{24}), the beakers were stirred again and gas samples were taken by syringe from the headspace of the beaker headspace (Wang *et al.* 1999). Ten millilitres of gas in the syringe was then injected into an auto-sampler vial. The concentrations of CH_4 and N_2O in the vial were simultaneously analysed with a gas chromatograph equipped with flame ionization detector (FID) and electron capture detector (ECD). The incubation lasted 57 days. The Eh in the slurry was monitored and measured with a pH-Eh meter connected to a platinum electrode.

Production potential of CH_4 or N_2O were calculated based on the equation from Lantin *et al.* (1995) as follows:

$$E = (C24 - C0)x\ \frac{Vh}{20\ g}\ x\ \frac{mW}{mV}\ x\ \frac{273}{(273 + T)}$$

E : CH_4 or N_2O production (mg CH_4 or N_2O g^{-1}soil)

C_0 : CH_4 or N_2O concentration in time 0 (ppm)

C_{24} : CH_4 or N_2O concentration after 24 hours (ppm)

*V*h : Volume of headspace of beaker (ml)

*m*W : Molecular weight of CH_4 or N_2O (g)

*m*V : Molecular volume of CH_4 or N_2O (22.41 litre at *stp, standard temperature and pressure*)

T : Temperature of incubator (^oC)

The total CH_4 and N_2O production potential were calculated as follows:

$$\text{Total } CH_4 \text{ or } N_2O \text{ production } = \sum_i^n (RixDi)$$

Where R_i is the production potential rate of CH_4 or N_2O (mg g^{-1} soil $hour^{-1}$) in the i^{th} sampling interval, D_i is the number of days in the i^{th} sampling interval and n is the number of sampling intervals.

2.2.4. Statistical analyses

The data were analysed using SAS software (SAS Institute 2003). The mean comparison between treatments was established by Tukey HSD test. Simple and multiple correlation analysis between total production of CH_4-N_2O and soil parameters was established to find out the effect of water treatments on production CH_4-N_2O from different soil types.

2.3. Results and discussion

2.3.1. Methane production

The time course of CH_4 production differed between the soils (Figure 2.3). Early in the experiment, the CH_4 productions were high in all of the treatments, likely occurs because of all soils produced CH_4 immediately after submergence (indicated by a flush). The production potential of CH_4 slightly decreased in second measurement. The drainage in FDF treatment was conducted twice. Drainage was started 15 and 36 Days of Incubation (DOI), but then the soils were re-flooded at 20 and 41 DOI. Watanabe *et al.* (2010) reported that wetting and drying of the soil could change the composition, population and transcriptional activities of the methanogenic archaea. Before drainage *Rice cluster I, Methanomicrobiales* and *Methanosarcinales* exist, but after drainage only the Methanomicrobiales were detected (Sugano *et al.* 2005). It means that the CH_4 is still produced in flooded or in drained condition, the difference is only amount of CH_4 production. During the measurement period, there were different peaks of CH_4 production in each of the soil likely occurs because of the existence of methanogenic bacteria. According to Roy *et al.* (1997), the reason for different initiation time of CH_4 production in different soils likely occurred because of the difference in abundance of viable methanogenic archaea in the air-dried. Unfortunately, in this study did not observe the activity of microorganism.

In all cases, the CF treatment had the highest CH_4 production compared to the FDF treatment. There were huge amounts of CH_4 degassing from FDF treatment when the water was drained, therefore CH_4 production from FDF were comparably low. Somehow, the trend of CH_4 production between CF and FDF treatments almost similar probably due to effect stirring of the soil. The ratio between soil and water should be carefully adjusted based on the soil texture. More clay content in the soil reduced the velocity of the magnetic bar inside of the soil. The velocity of magnetic bar is important aspect in this incubation experiment because stirring can release the gas that entrapped in micro-pore of the soil and purify the gas in the micro-pore of the soil with wash out the soil using N_2. Based on the procedure for gas sample collection are described by Mitra *et al.* (2002) that incubation experiments were conducted by placing a magnetic bar inside the incubation bottle to stir the soil during gas sampling to release gas which entrapped between the micro-pore of the soil. Besides release gas, soil stirring also increase the chance for oxygen diffusion into soil. The oxygen concentration influence the production of CH_4 in both treatments. So, although in CF or FDF, both of the treatments could release the gases which entrapped between the soils. The CH_4 production in the CF treatment from Klaten, Boyolali, Grobogan, Demak and Pati ranged approximately 0.06-1.32; 0.08-0.88; 0.11-1.30; 0.06-0.88 and 0.05-1.33 mg C g^{-1} soil, respectively. However, the CH_4 production in the FDF treatment from Klaten, Boyolali, Grobogan, Demak and Pati ranged approximately 0.07-0.74; 0.04-0.88; 0.09-0.78; 0.08-0.60 and 0.02-0.64 mg C g^{-1} soil day^{-1}, respectively.

Figure 2.4 shows the cumulative CH_4 production in each of the soils. This figures show that the first water drainage resulted the lower CH_4 production in all soil. Compare to CF treatment, the FDF treatment in Klaten, Boyolali, Grobogan, Demak and Pati's soils reduced the cumulative of CH_4 production by approximately 23.7; 21.6; 34.7; 11.4 and 21.9%, respectively. This result is similar to the finding in paddy field studies reported by Cai *et al.* (2003); Kang *et al*. (2002); Zhang *et al*. (2011), i.e., that CH_4 fluxes were higher during flooded condition rather than drained during the fallow condition. High CH_4 production in CF may have caused by soil water conditions. It is one of the factors that control CH_4 production in paddy soil, due to methanogenesis takes place under strict anaerobic reducing conditions. FDF treatments improved soil aeration and facilitated O_2 diffusion from the atmosphere into the soil. This condition inhibit the formation of CH_4 due to O_2 availability is the major factor limiting methanotrophy bacteria. King *et al.* (1990) proved the importance of O_2 availability in Florida swamps where gas diffusion is easy, therefore methanotrophyc bacteria was significant in peat.

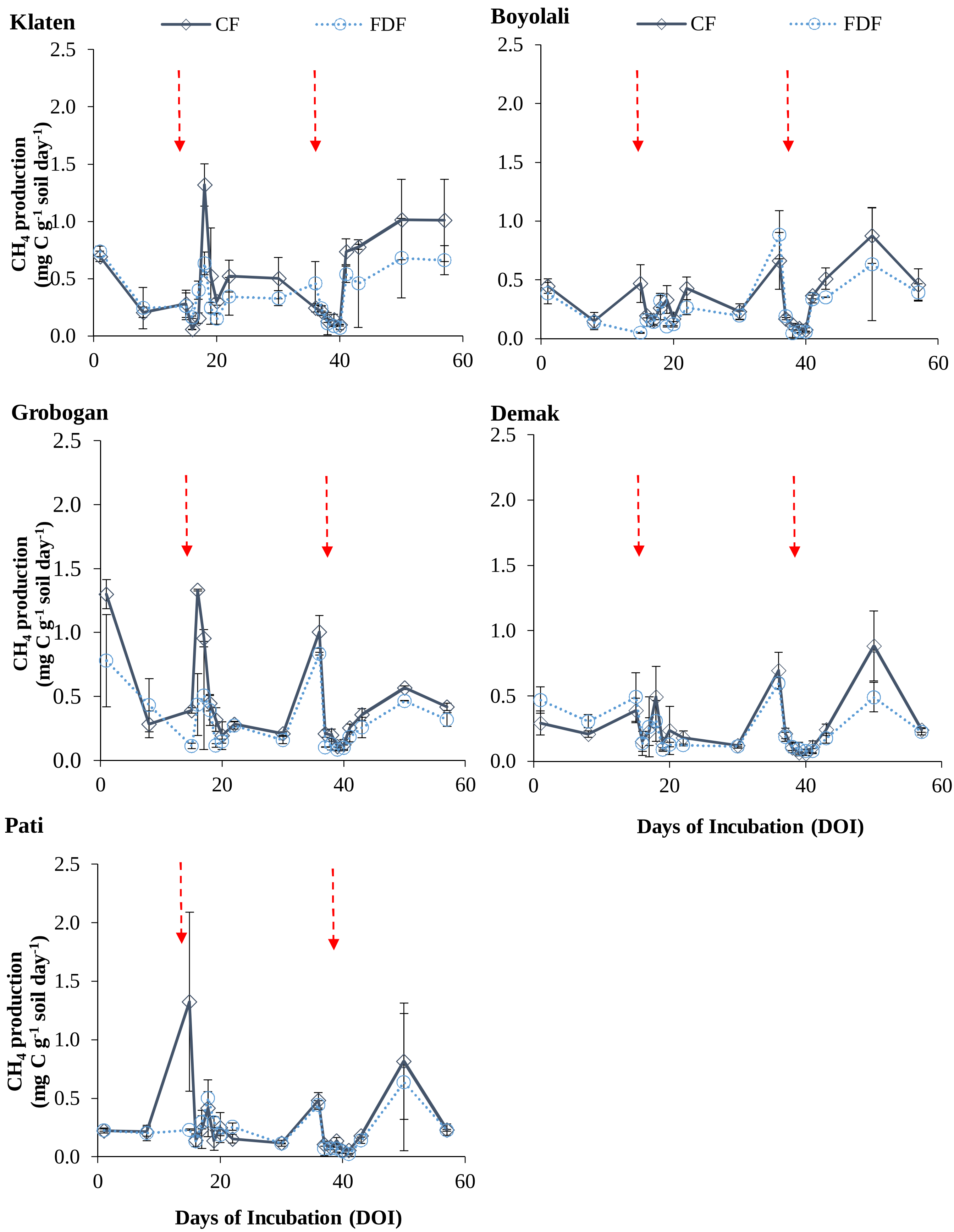


Figure 2.3. The time course of CH_4 production rate during soil incubation. Red arrow means drainage the water in FDF treatment

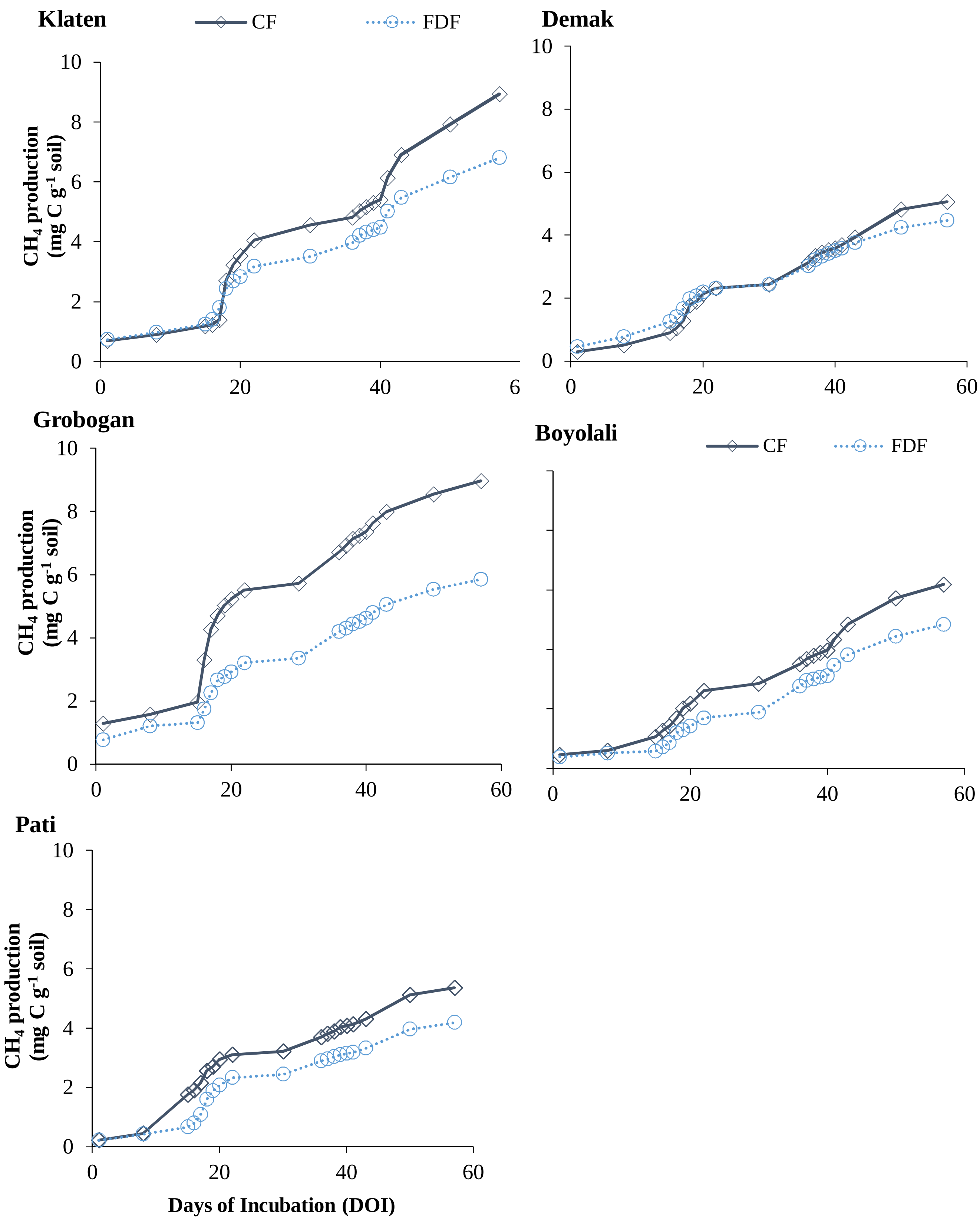


Figure 2.4. The pattern of cumulative CH_4 production during soil incubation

Although after re-flooded, the CH_4 production from FDF treatment almost in all soils were lower than CF treatment. Li *et al*. (2011) reported that drainage in longer period reduces oxidants in the soil, therefore soil redox (Eh) drop to favourable level for CH_4 production and it is difficult for methanogenesis to survive and CH_4 emission becomes lowered. The duration of the reduction processes varied greatly between the soils. According to Yao *et al.* 1999, NO_3^- was reduced first, then reduction of Fe(III) and SO_4^{2-} were completed later.

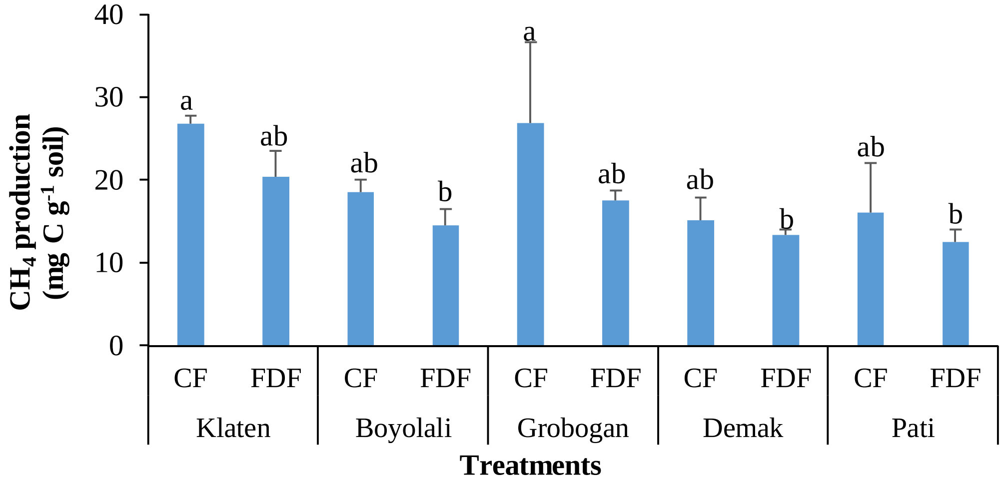


Figure 2.5. The total of CH_4 production during soil incubation, $P < 0.05$

There were significant effect of different soil types and water treatments on the CH_4 production. The highest CH_4 production was from Grobogan soil under CF treatment approximately 26.92 mg C g^{-1} soil and the lowest CH_4 production was from Pati under FDF treatment approximately 17.57 mg C g^{-1} soil (Figure 2.5). Klaten's soil produced highest CH_4 followed by Grobogan, Boyolali, Pati and Demak's soil were approximately 23.63; 22.24; 16.53; 14.33 and 14.29 mg C g^{-1} soil, respectively. The CH_4 production from Klaten. Grobogan and Boyolali's soil higher compare to Demak and Pati likely occurred because of the chemical characteristic of the soil. Klaten. Grobogan and Boyolali's soil have high C and N content in the soil. Similar observation were reported by Inubushi *et al.* (1990) and Kimura (1992). Those studies showed that there was positive correlations between the amounts of CH_4 formed in paddy soils and several soil parameters such as the content of C and N. Soil organic matter related with Eh and provide soluble C which favour the formation of both CH_4.

2.3.2. Nitrous oxide production

The patterns of N_2O production rate from five paddy soils were quite different from those of CH_4 production (Figure 2.6). The N_2O production fluctuated during soil incubation and the pattern for each treatment were quite different. There was no consistent change in the rate of N_2O production from CF and FDF treatments. In FDF treatment, sometimes peaks of N_2O production occurred after re-flood. This finding was similar with Beare *et al.* (2009) that reported N_2O production in soils was reduced by 93–96% during the drainage phase and the majority (88%) of the N_2O production occurred after re-flooded from compacted soil. However, effect of drying and re-wetting the soil was inconsistent with those found in previous studies. Cai *et al.* (1997, 2001) reported that peak of N_2O appeared at the beginning of the disappearance of floodwater in rice fields. The production of N_2O in the FDF treatment from Klaten, Boyolali, Grobogan, Demak and Pati ranged approximately 0.12-31.27; 0.12-3.94; 0.10-14.18; 0.05-3.26 and 0.22-3.00 mg µg N g^{-1} soil, respectively. However, the production of N_2O in the CF treatment from Klaten, Boyolali, Grobogan, Demak and Pati were approximately 0.28-29.32; 0.09-13.73; 0.12-15.30; 0.04-6.84 and 0.09-2.46 µg N g^{-1} soil, respectively.

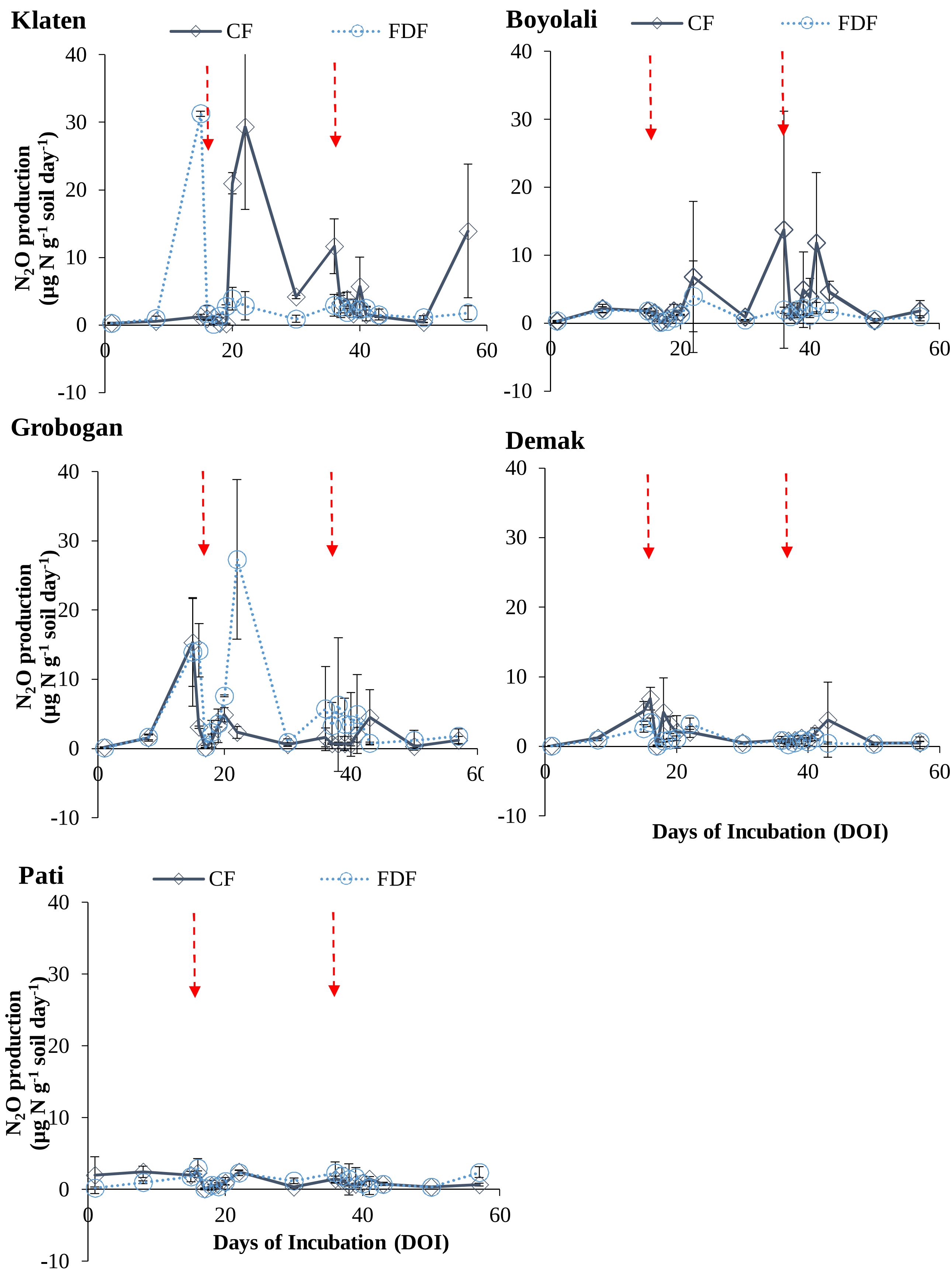


Figure 2.6. The time course of N_2O production rate during soil incubation. Red arrow means drainage the water in FDF treatment

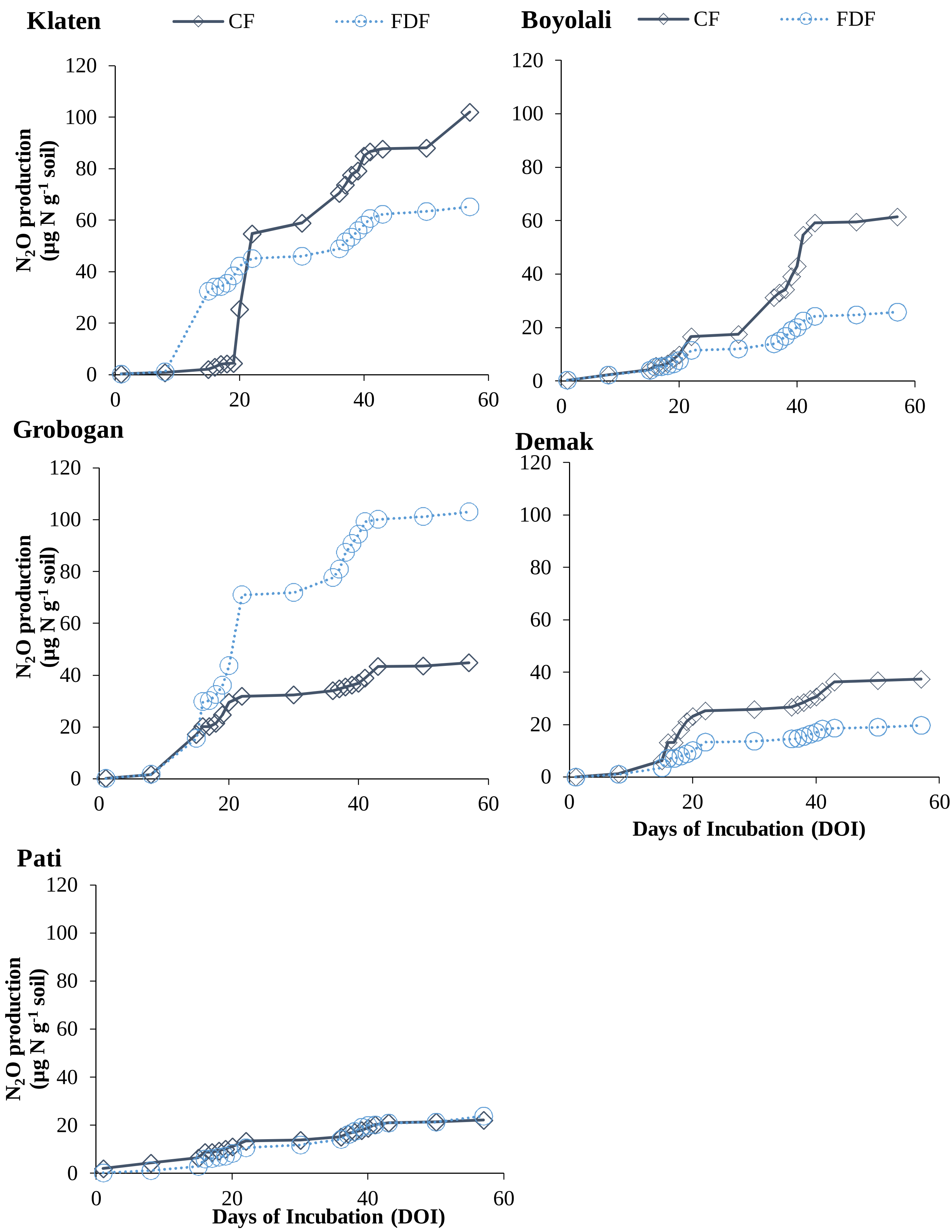


Figure 2.7. The pattern of cumulative N_2O production during soil incubation

Figure 2.7 show that the N_2O production reached stationary phase in almost all soil in 43 day of incubation (DOI). Drainage stimulated cumulative production of N_2O only in Grobogan and Pati's soil, likely occur because FDF triggers interchangeable nitrification of ammonia, denitrification of nitrate and nitrifiers produced more N_2O than denitrifiers. Cai *et al.* (1997) and Zou *et al.* (2005) found that alternate flooding and drying cycle considerably increased N_2O emission. The highest N_2O production rate by nitrifiers was observed at 90 % WHC, when the soil had become partly anaerobic, as indicated by the high denitrification rate (Klemedtsson *et al.* 1988). While N_2O production was lower in Klaten, Boyolali and Demak's soil due to the drainage. During the CF period, less N_2O emission occurred as nitrification process was inhibited by flooding water and N_2 was the main product of denitrification; also N_2O transport was retarded in water saturated soil (Freney and Denmead 1992; Granli and Bockman 1994).

Although there were very large variations in the N_2O production during soil incubation, there was interaction between soils and water treatments. The production of N_2O from the FDF treatment in Klaten, Boyolali and Demak's soil were lower than those from the CF treatment approximately 36.1; 58.0 and 46.9%, respectively. FDF treatment resulted in large increased in N_2O production relative to the CF treatments on Grobogan and Pati's soil approximately 130 and 7.6%, respectively, but in Pati's soil showed no significant difference between both treatments. On the other hand, FDF treatment stimulated N_2O production in Grobogan's soil likely occur because there was one big peak during 22 DOI in FDF treatment. This peak likely occur because the Eh reached the lowest value at that time. This condition could be happened likely occur because of Grobogan's soil contain high clay. When the soil was stirred the stirrer cloud not stir properly due to heavy soil. Therefore, the gas that entrapped in soil micro-pore cannot release and accumulated until the next gas sampling. Although N_2O is a by-product of nitrification and an immediate product of denitrification, the different effect on N_2O production due to water treatment in soil likely occurs because of length of incubation and differences in soil moisture of different soil types which affect N_2O production (Klemedtsson *et al.* 1988).

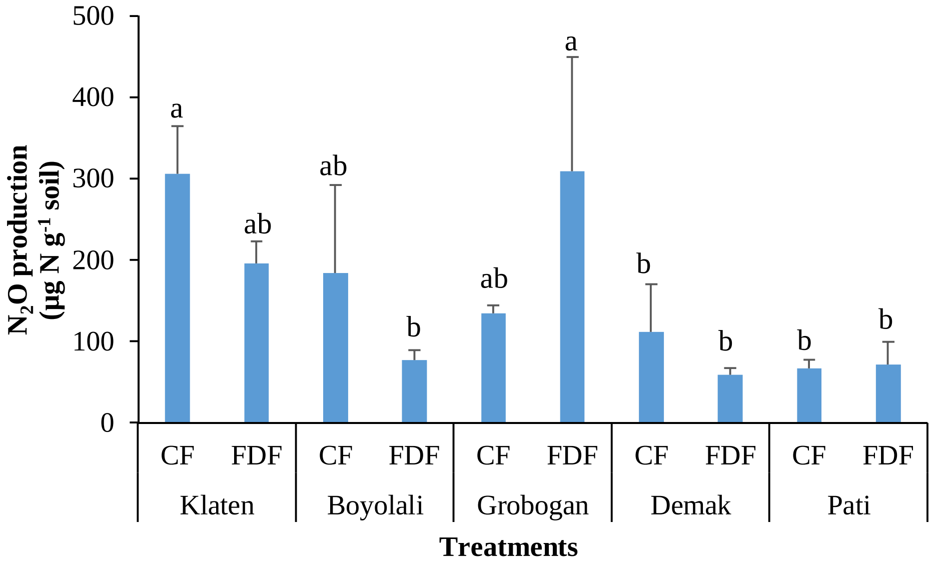


Figure 2.8. The total of N_2O production during soil incubation, $P < 0.05$

Nitrous oxide patterns were greatly affected by soils (Xiong *et al.* 2007). The N_2O production from incubation of Klaten, Boyolali, Grobogan, Demak and Pati's soil were approximately 251; 130; 221; 85 and 69 µg N g^{-1} soil, respectively (Figure 2.8). Klaten's soil resulted the highest N_2O production probably because the high of total N in Klaten's soil. According to Baggs *et al.* (2000) that more N will be available for nitrification and denitrification processes and higher N_2O emissions may occur.

2.3.3. Soil redox potential

The difference of soil Eh between CF and FDF in each of the soil can be seen clearly in Figure 2.9. In all the soil FDF treatments could reach higher soil Eh compare to CF. Soil Eh from CF treatment in Klaten, Boyolali, Grobogan, Demak and Pati's soil were approximately -238 to 156; -410 to 76; -163 to 54; -308 to 52 and -228 to 1 mV. However, Eh from FDF treatment in Klaten, Boyolali, Grobogan, Demak and Pati's soil ranged approximately 146 to 164; -256 to 13; -145 to 97; -166 to 46 and -227 to 55 mV. Wang *et al.* (1993) and Neue *et al.* (1994) reported that rice field management and indigenous soil characteristics, such as Fe_2O_3, SO_4, MnO_4, silt and carbon content, affect the potential CH_4 production of soils. These properties could affect the redox status of the soils in a reduced condition; which in turn may influence to the production of CH_4 by methanogenic bacteria. A rapid decrease in Eh after flooded due to high carbon content in soil clay, i.e, in Klaten, Boyolali and Grobogan, appears to explain the greater CH_4 production potential.

Soils like Boyolali with high C content but low active Fe content attain Eh values less than −200 mV soon after submergence. Similar observation was reported earlier by Ponnamperuma (1972, 1981). Methanogenesis occur under strictly anaerobic condition. In this study, it can be seen clearly that Eh in CF lower than FDF, especially in Grobogan's soil (Figure 2.10). The low of soil Eh made CH_4 production potential higher in Grobogan soil. A sufficient low redox (Eh) potential is required for CH_4 production. Once the soil is flooded, organic matter starts decomposing accompanied with a stepwise biochemical reduction of the soil which is indicated by a lowering of the redox potential (Ottow 1981; Inubushi *et al.* 1984: Neue 1985). Soil Eh in water-logged condition is primarily related to the amount and kind of organic matter in soil (Ponnamperuma 1972). Soil Eh was found negatively correlated with CH_4 production in and emission from the flooded soil. However, Eh development alone may not be a good indicator for the onset of methanogenesis and should only be used when the soil and its CH_4 production behavior have been characterized (Yagi *et al*. 1996; Sigren *et al*. 1997; Yao *et al*. 1999). Soil Eh is also an important factor affecting N_2O emissions from paddy fields.

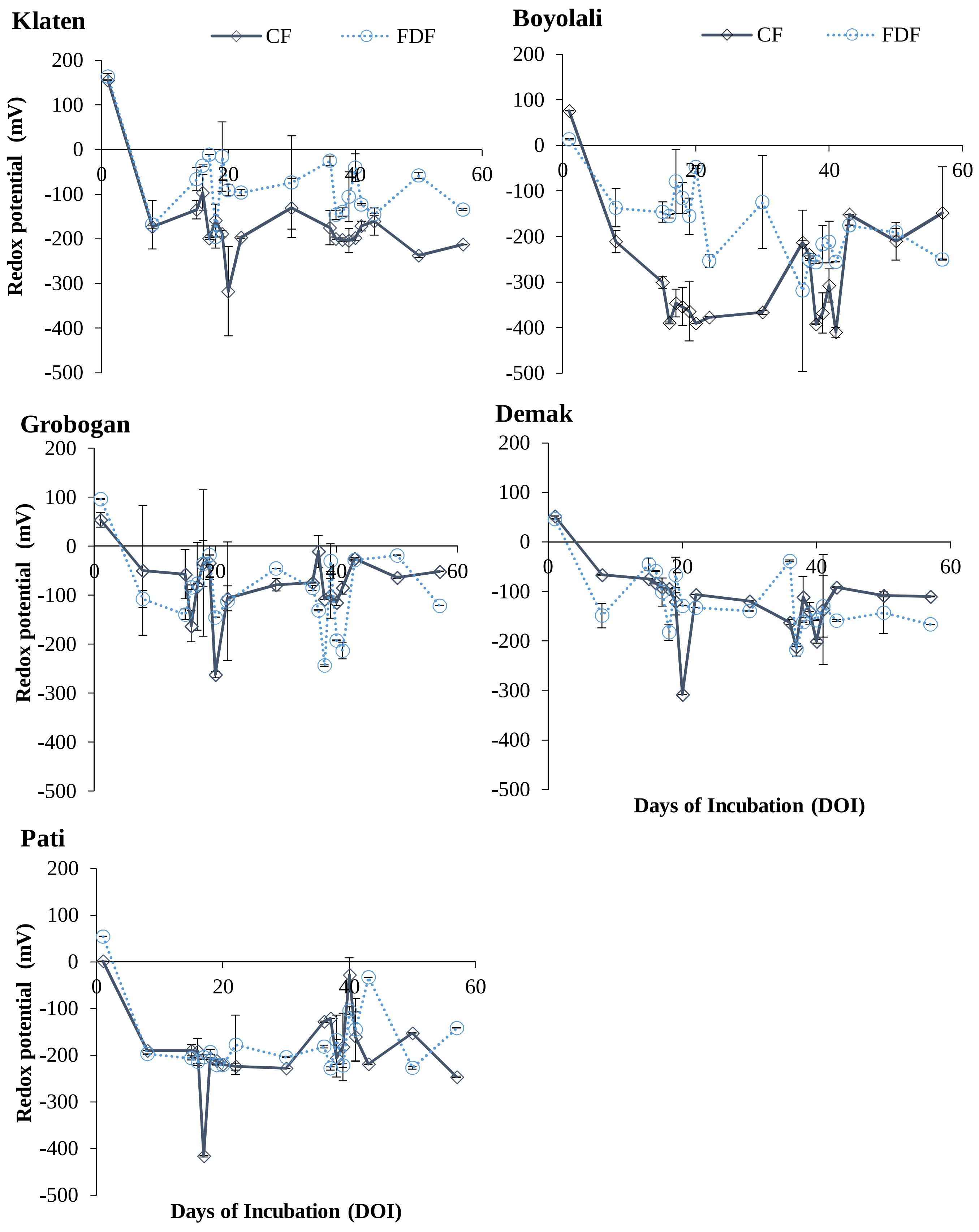


Figure 2.9. Redox potential during soil incubation

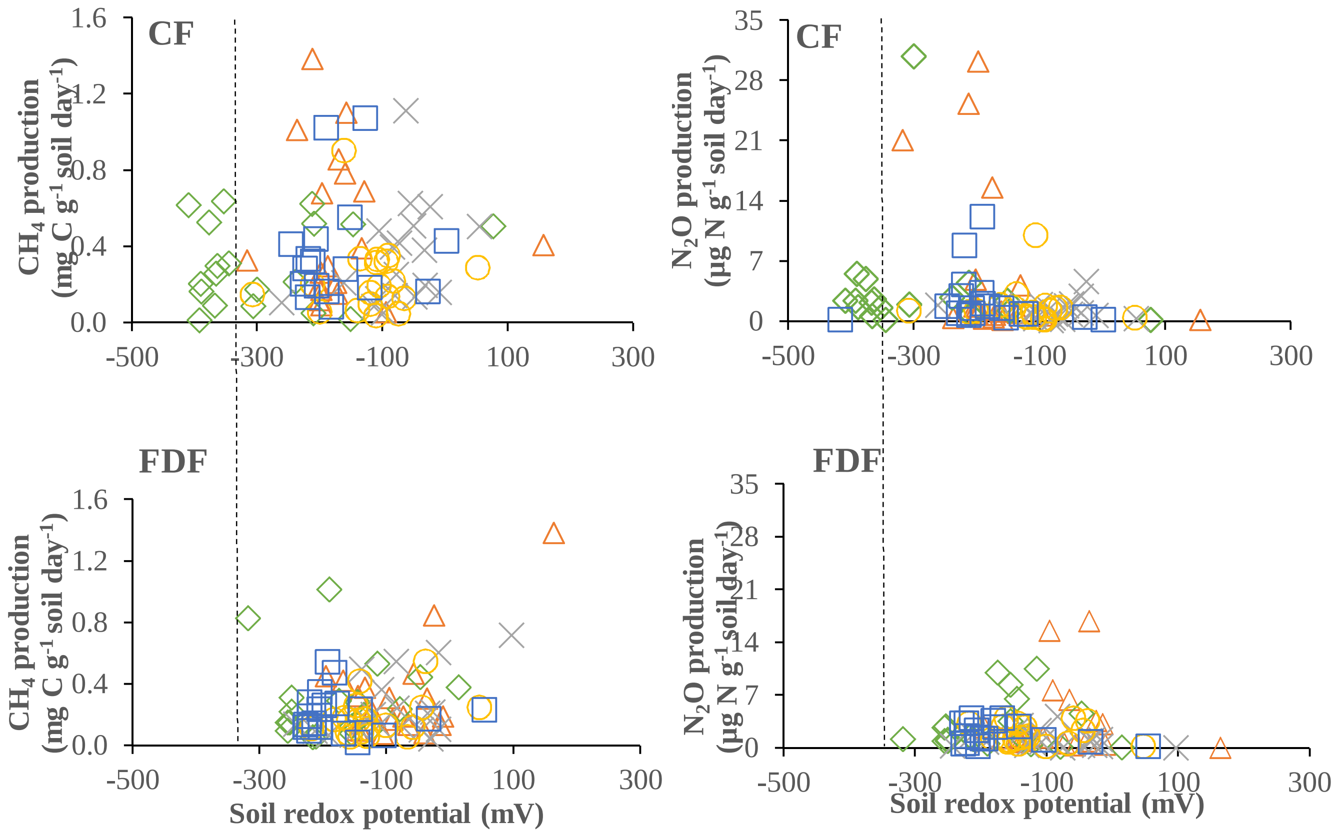


Figure 2.10. Correlation between CH_4-N_2O and soil Eh during soil incubation

2.3.4. Effect of soil properties on CH_4-N_2O emission

CH_4 production potentials showed pronounced variations among the different soils. Results of simple regression correlation analysis between different soil properties and CH_4-N_2O production indices are presented in Table 2.2. Soil properties like soil organic carbon, NO_3^-, NH_4 and total Fe significantly affected the CH_4 production potentials. However, NO_3^-, total Fe and total Mn significantly affected the N_2O production potentials. In this study, simple regression correlation analysis showed that soil organic C had significant effect on CH_4 production. It could be due the dominating acetoclastic pathways for CH_4 production rather than hydrogenotropic pathway under waterlogged incubation (Conrad and Klose 1999). Wang *et al.* (1993) also observed no correlation between soil organic C and CH_4 production. Earlier study, Inubushi *et al.* (1990) and Kimura (1992) observed positive correlations between the amount of CH_4 formed in paddy soils and several soil parameters such as the content of organic-C, water-soluble organic-C and mineralizable-N. The content of soil organic C lower the Eh and provide soluble C which favour the formation of both CH_4 and N_2O. However, Yagi *et al.* (1990), on the contrary, found no correlation between CH_4 production rates and total C contents in soils.

Table 2.2. Correlation between soil characteristics and CH_4-N_2O production under CF and FDF treatments

Variables	r values	
	CH_4	N_2O
Soil organic carbon	0.443**	0.235
SiO_2	0.156	0.012
P_2O_5	0.055	0.093
NO_3^-	-0.404**	0.426**
NH_4^+	0.355*	0.093
Total Fe	-0.404**	0.584**
Total Mn	-0.292	0.337*
SO_4^-	0.124	0.222

**P*<0.05, ***P*<0.01, without * or **: not significant.

The presence of NO_3^- in the soil is one of the important factors controlling CH_4-N_2O production. It has been demonstrated that existence of NO_3^- can inhibited CH_4 and stimulated N_2O production. The presence of NH_4 in the soil is also one of the important factors controlling the CH_4, NH_4 can stimulate CH_4 emission from rice paddy fields due to the competition of NH_4 for the oxidation with CH_4 by methanotrophs (Mosier *et al.* 1991). The NH_4 leads to an increase in nitrified population relative to methanotrophs and thus the overall CH_4 oxidations reduces, as nitrifiers oxidize CH_4 less efficiently than methanotrophs (Willson *et al.* 1995). In the study there was correlation between CH_4-N_2O production and active Fe and Mn contents. The finding is synergy with those of Wang *et al.* (1993) who reported that high Fe and Mn in soils inhibited CH_4 production. Takai and Wada (1990) also postulated that the content of bio-active Fe may be the most important controlling factor in CH_4 production. Iron is an important oxidant for biological and chemical reactions that use oxidizing or reducing agents. Nitrite can be reduced by the presence of iron oxide at a near-neutral pH, and the end product is N_2O and NO as an intermediate (Van Cleemput and Baert 1983; Van Cleemput 1998). In this study there was no significant correlation between CH_4 production and P_2O_5 (available P). The findings do not comply with those of Adhya *et al.* (1998) who reported that addition of K_2HPO_4 in soil had a stimulatory effect while Mussorie rock phosphate and single super phosphate had an inhibitory effect on CH_4 production, due to the presence of sulphur (S) in them.

2.3.5. Global warming potential and contribution of each gas

The range of total GHG emission from CF and FDF treatment were approximately around 11-21 and 9-15 mg CO_2 eq g^{-1} soil, respectively (Table 2.3). The highest GHG emission reduction was found in Boyolali followed by Klaten, Pati, Demak and Grobogan, were approximately around 30, 27, 19, 18 and 14%, respectively. High GHG emission reduction in Boyolali and Klaten's soil likely occur because high soil carbon. The content of soil organic C lower the Eh and provide soluble C which favour the formation of both CH_4 and N_2O. The difference of Eh between CF and FDF treatments in Boyolali and Klaten were wider than the difference of Eh in other soils. Inubushi *et al.* (1990) and Kimura (1992) found positive correlations between the amount of CH_4 formed and the content of organic-C in paddy soils. To evaluate the climate implication of the cultivation practices, it is desirable to have relative contribution of each gas to global warming. In this study, CH_4 emission from different water treatments account for 67-90% of the contribution to global warming, while N_2O emission is only contributed 10-33%. Water treatments influence GHG emission by changing soil water content, which determines aerobic and anaerobic conditions in the soil. Aerobic and anaerobic conditions were related with soil redox potential (Eh), which has been used as one of the most indicative soil parameters for CH_4 and N_2O from irrigated rice fields (Hou *et al.* 2000).

Table 2.3. Global warming potential and contribution of each gas

Soil name	Water treatments	CH_4	N_2O	GHG emission	Reduction	Contribution each treatment to GHG emission CH_4	N_2O
		mg CO_2eq g^{-1} soil			(%)	(%)	
Klaten	CF	16	5.0	21		76	24
	FDF	12	3.2	15	27	79	21
Boyolali	CF	11	3.0	14		78	22
	FDF	8	1.3	10	30	87	13
Grobogan	CF	16	2.2	18		88	12
	FDF	10	5.1	15	14	67	33
Demak	CF	9	1.8	11		83	17
	FDF	8	1.0	9	18	89	11
Pati	CF	9	1.1	11		90	10
	FDF	7	1.2	9	19	86	14

2.3.6. Contribution of soil to national emissions

According to soil maps from ICALRRD that fifth of paddy soil from Klaten, Boyolali, Grobogan, Demak and Pati district, were classified as Inceptisols. Inceptisols is the largest paddy soil in Indonesia and the area was approximately around 59.69 million ha or around 32% of Indonesian's paddy soil (ISRI 2006). The average of CH_4 and N_2O production from fifth soils were approximately 11.37 g and 2. 66 g CO_2 eq ha^{-1} $year^{-1}$, respectively. Total emission from Indonesia for CH_4 and N_2O ranged approximately 236,388 and 28,341 Gg

CO_2eq (SNC 2010). Based on the area and the average of CH_4 and N_2O production from Inceptisols, therefore Inceptisols contributed to national CH_4 and N_2O production were approximately 0.68 and 0.16 Gg CO_2eq, respectively (Table 2.4).

Table 2.4. National emission (SNC 2010) and contribution of soil to national emissions

	National emission	**Inceptisols**	**Contribution**
Gases	**Gg CO_2eq**		**%**
CH_4	236,388	0.68	0.000287132
N_2O	28,341	0.16	0.000560551

2.4. Conclusions

Five rice soils from different locations in Central Java were incubated anaerobically for 57 days to determine methane (CH_4) and nitrous oxide (N_2O) production potentials and to establish relationships between chemical properties of soils and CH_4-N_2O production potential based on different water treatments. Compare to CF treatment, the FDF treatment in Klaten, Boyolali, Grobogan, Demak and Pati's soils reduced the CH_4 production by approximately 23.7; 21.6; 34.7; 11.4 and 21.9%, respectively. However, FDF treatment could not reduce N_2O production in all soil types. Thus, we need to be cautious while recommending a particular irrigation regime for rice cultivation in order to avoid substantial emission of one or the other greenhouse gas. Soil organic carbon showed significant correlation with CH_4 production while total Mn showed significant correlation with N_2O production. Total Fe, NO_3^- and NH_4^+ have significant correlation with CH_4 and N_2O production.

CHAPTER 3

EFFECT OF WATER MANAGEMENTS ON GREENHOUSE GAS EMISSION FROM PADDY FIELD IN INDONESIA

3.1. Introduction

In previous chapter 2, evaluating CH_4 and N_2O production under laboratory-scale has made clear that chemical characteristics of the soil is important factor that determine CH_4 and N_2O production from the soil. It has been observed that water treatment could reduce CH_4 production in different soil types. There is a trade-off between CH_4 and N_2O production in paddy soils. Somehow, reducing water from the soils stimulate N_2O production. Based on previous study in chapter 2, the experiment related with water treatments should be conducted in field which use indicator plant to determine how CH_4 and N_2O emissions from these types of treatments can be reduced and to examine how much water can be saved. In this chapter used one of the soils that was observed in previous chapter.

Rice field is an important source of CH_4 and N_2O, but it can also be a sink for CO_2 through C sequestration into biomass products and soil organic matter (Johnson *et al.* 2007). Miyata *et al.* (2000) observed that net CO_2 flux from the rice paddy significantly larger when the field was drained than when it was flooded due to no diffusion barrier by the floodwater. While according to Alberto *et al.* (2009), soil CO_2 efflux is reduced due to limitation of diffusion of oxygen and suppression of CO_2 emissions in flooded fields. Ruser *et al.* (2006) found that CO_2 production from a fine-loamy soil fertilized with nitrate was not strongly effected by soil moisture. Liu *et al.* (2013) reported that there was a negative rate of CO_2 flux in the daytime and a positive throughout the night most likely because during the daytime plant photosynthesis uptake of CO_2 from both the atmosphere and from respired CO_2 emitted by the soil and floodwater. Respiration at night leads to an efflux of CO_2 to the atmosphere. In this study, gas samples were taken in the morning and soil type in field experiment is loam, therefore CO_2 emission from this study was omitted.

Many cultivation practices have been improved in order to decrease environmental burdens of paddy rice production and to improve yield potential. The concern about the sustainability of food production has increased because of the increasing of world population and the growing of environmental issues. Due to increasing scarcity of freshwater resources, water-saving regimes are needed to reduce water use from rice field. Alternate wetting and drying (AWD) is a method in irrigated rice cultivation to save water, the rice fields are allowed to dry intermittently but still adequately supplied with water for the roots of rice plant to maintain rice yields. AWD is conducted by drying and re-flooding of the rice field and the time intervals between dry and wet conditions appear to be too short to facilitate the shift from aerobic to anaerobic soil conditions (Wassmann *et al.* 2000). Groundwater table are used in these system. The ponded water on the field is allowed to drop to 15–20 cm below the soil surface before irrigation is applied (Rejesus *et al.* 2011). On the other hand, oxic/anoxic boundary of the soil has important effects on GHG production (Dinsmore *et al.* 2009). CH_4 emissions are high under strictly anaerobic conditions (Moore and Dalva 1993), while N_2O emissions are high in intermediate conditions (Davidson *et al.* 2000). It has been reported that mid-season drainage could mitigate CH_4 emission conversely it could lead to an increase in N_2O emission (Bronson *et al.* 1997). Flooded rice fields are not a potent source of atmospheric N_2O because N_2O is further reduced to N_2 under the strong anaerobic conditions (Granli and Bockman 1994).

Simultaneous mitigation options are different for CH_4 and N_2O emission, and minimizing one gas may increase the emission of the other, since the production of these two gases take place under contrasting conditions (Ghosh *et al.* 2003). So, the trade-off both of the emission should be well prepared for a balanced set of mitigation options, which optimize the emission trade-off in minimum cumulative radiative forcing of the two gases on global warming potential (GWP), thus having a lowest possible greenhouse effect. On the other hand, this option have to be carefully sorted out if the mitigation option should not come in the way of achieving high crop yields. The objective of this chapter is to investigate the effects of AWD, site specific AWD and continuous flooded on CH_4-N_2O emission, yield and water productivity from paddy field in Indonesia

3.2. Material and methods

3.2.1. Site description

The experimental field was located at the experimental farm of Indonesian Agricultural Environment Research Institute (IAERI), Jakenan (06^{0}46'42.1" S, 111^{0}11'52.0" E), in the Pati district, Central Java, Indonesia. The soil type at IAERI experimental farm is an Inceptisols (Aeric endoaquept) classified by USDA (2014). The physicochemical characteristics of the soil are listed in Table 3.1. The soil pH is 5.6 and soil texture is medium loam. The total carbon and nitrogen contents of the soil are 1.6 and 0.2 g kg^{-1}, respectively.

The study were conducted during the rainy season (RS) 2014 which ran from March to June 2014. According to meteorological data that was collected from IAERI weather station, the mean annual air temperature in 2014 was 27.7°C, and the mean air temperature during March to June 2014 was 31.3°C. The annual rainfall in 2009 was 2000.3 mm, and the rainfall during March to June 2014 was 365.50 mm (Figure 3.1).

Table 3.1. Physicochemical characteristics of the experimental soil

Parameters	Values
Texture (%)	
Sand	43
Silt	40
Clay	17
pH	
H_2O	5.6
KCl	4.9
EC (dS/m)	0.035
Total (g kg^{-1})	
C	1.6
N	0.2
Available P (mg kg^{-1})	4.8
Available K (mg kg^{-1})	14.1
1M NH_4OAc extractable ($cmol^+$ kg^{-1})	
Ca	10.98
Mg	0.85
K	0.03
Na	0.13

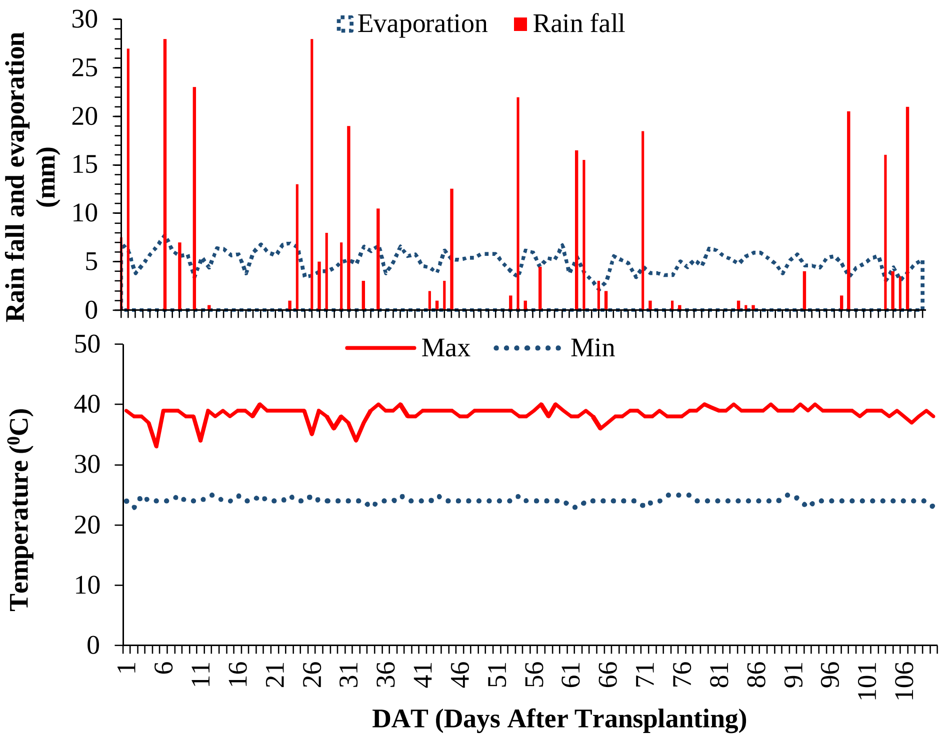


Figure 3.1. Meteorological data during the experimental period RS

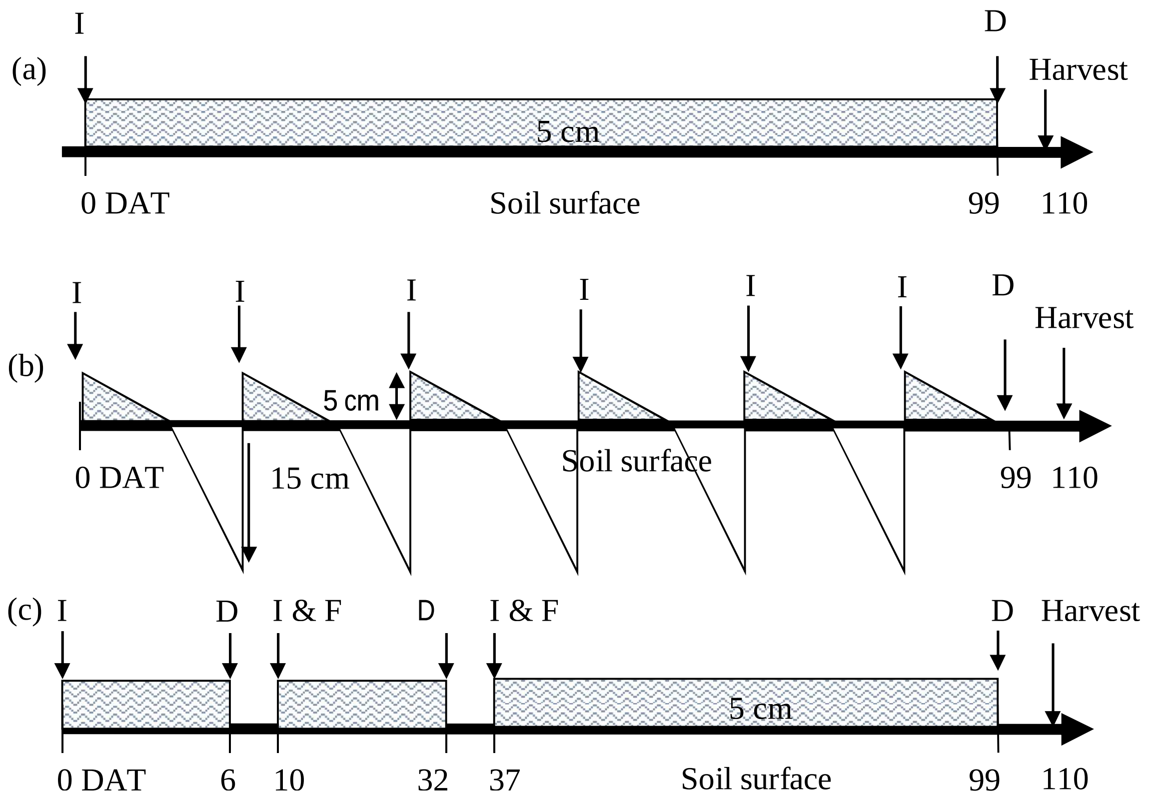
3.2.2. Experimental design and culture practices

The crops were established by transplanting, fourteen-day-old seedlings of the Cisadane rice variety were transplanted into each 5 m x 7 m plot, with 20 cm x 20 cm plant spacing and two-three seedling per hill. The physiological and agronomic characteristic of Cisadane is shown in Table 3.2. The days from sowing to harvest for this variety are 135 to 140 days. The fields were plowed and puddled thoroughly to a 10 cm depth 5 days before transplanting. Every experimental plot received fertilization at rates of 120 kg N ha^{-1} (urea), 60 kg P_2O_5 ha^{-1} (super phosphate) and 90 kg K_2O ha^{-1} (potassium chloride). Super phosphate at 60 kg ha^{-1} was applied 1 day before transplanting as the basal dose. Urea and K_2O fertilizers were broadcast as three split applications at rates of 40 and 30 kg ha^{-1} for each application, respectively. Urea and K_2O fertilizers were applied 11, 38, and 56 days after transplanting (DAT).

Table 3.2. Physiological and agronomic characteristic of Cisadane

Parameters	Cisadane
Date release	18-Feb-80
Origin	Pelita I-1/B2388
Group	Cere (indica)
Growth period	135 – 140 days
Plant height	105 – 120 cm
Productive tillers	15 – 20 hills
Weight per 1000 grains	29 g
Productivity	5 Mg ha^{-1}
Yield potential	7 Mg ha^{-1}
Plant shape	Straight
Foot colour	Green
Auricle colour	Colourless
Leaf tongue colour	Colourless
Leaf colour	Green
Leaf surface	Coarse
Leaf position	Straight
Grain colour	Yellow
Amylose content	20%

The experiments were arranged in a randomized block design with 3 treatments replicated three times. The plots were comprised of water management: continuous flooding (CF), alternate wetting and drying (AWD) and site specific of AWD (S-AWD) (Figure 3.2). Piezometer was made from PVC pipes with 4 cm in diameter and 100 cm in length. Piezometer were installed in the field keeping 20 cm above the soil and the remaining 80 cm which was perforated underneath to measure the depletion of soil water in the field. Each of the plot was installed 2 pieces of piezometer. Water irrigation was controlled every day. In CF treatment, standing water was kept 5 cm above soil surface until 2 week before harvest. In AWD treatment, irrigation water was applied when depleting water table inside the pipe reached a 15 cm below soil surface. However, in S-AWD the water allowed to drain 1 week before first and second fertilization as long 7 days. The inner borders of the experimental plots were lined with plastic sheets up to 40 cm soil depth to prevent lateral water flow between plots.



Noted: I = irrigation, D = drainage and F = fertilization

Figure 3.2. Schematic diagram of different water managements (a) CF, (b) AWD and (c) S-AWD

3.2.2. Measurement of yield and yield component

Plants were harvested when they completely matured. The plants were harvested on 107 DAT. The harvested area of each plot was approximately 3 x 3 m for determination of yield per unit area. The information related to yield and all the yield component, i.e., plant height, effective tillers, length of the panicle, number of spikelet per panicle, number of filled and unfilled grains per panicle, 1000 grain weight, grain yield, straw yield were collected and harvest index were calculated. Grain yield was adjusted to 14 % seed moisture content. The biomass was dried at 70°c for 48 hours. According to Fageria *et al.* (2011), the grain harvest index was calculated by using the following formula:

$$Grain\ harvest\ index = \frac{(Grain\ yield)}{(Grain\ yield + rice\ straw)}$$

3.2.3. Measurement of water saving and water productivity

Water discharge from the irrigation pipe was calculated as the volume of water (m^3) flowing through the pipe and measured as cubic meter per second ($m^3\ s^{-1}$). The time required to maintaining appropriate water levels in the main plots during each irrigation was noted and summed to calculate the total volume of water applied to the plots throughout the cropping season. The percentage of water saving was calculated as follows:

$$Water\ savings\ (\%) = \frac{Water\ supply\ in\ flooded\ plot - water\ supply\ in\ AWD\ plot}{Water\ supply\ in\ flooded\ plot}$$

Furthermore, water loss was also calculated based on the amount of water supplied in each plot. A simple measuring scale was used to determine the level of water (cm) lost each day during wetting period. Water Productivity Index (WPI) is water-use efficiency is intrinsically ambiguous in relation to crop production (Sharma 1989; Bhuiyan *et al.* 1995). WPI is calculated as the ratio of crop yield (kg h^{-1}) per unit water (m^3 h^{-1}) supplied as defined by Jaafar *et al.* (2000). It includes irrigation and rainfall.

$$Water\ productivity\ index\ (kg\ m^{-3})\ = \frac{Grain\ yield\ (kg\ ha^{-1})}{Total\ water\ supply\ (m^3 ha^{-1})}$$

3.2.4. Gas sampling

The CH_4 and N_2O fluxes were measured by closed chamber method and collected by using Plexiglas chambers during the rice-growing period (IAEA 1992). Each experimental plot had removable chambers for gas collection, which measured 50 cm x 50 cm x 100 cm for CH_4 sampling and 40 cm x 20 cm x 30 cm for N_2O sampling. Access to the chambers in paddy fields was provided by small footbridges to avoid gas bubbles/ebullition in the field. Four hills of rice plants were covered in each sampling CH_4 chamber, and no rice plants were covered in an N_2O chamber. There were 2 hills of rice plants that were left unplanted to leave space for N_2O chamber placement between the rice plants. Gas samples were collected once a week. However, when the plots of S-AWD was drainage, the gas sampling was collected every day. In each of the plot, gas sample of CH_4 was measured in triplicate, while N_2O was measured once. Five gas samples from each chamber of CH_4 and N_2O were collected with interval time at 0, 6, 12, 20 and 30 minutes started at 08:00 in the morning on each sampling day. Gas samples inside the chambers were collected using a 10-mL syringe fitted with a stopcock and transferred into 10 ml of vacuum vial then brought directly to the laboratory. The CH_4 and N_2O concentrations were directly analyzed with a gas chromatograph (GC), which was equipped with a flame ionization detector (FID) for CH_4 analysis and an electron capture detector (ECD) for N_2O analysis.

Soil pH and redox potential (Eh) were measured simultaneously with gas sampling. Platinum-tipped electrodes for determining the redox potential were inserted into the soil of each plot to a depth of 0.1 m and remained there for the whole rice-growing period. In each plot, the electrode was set up in four replicates by the quadrat after transplanting. The soil pH-Eh was measured by using a portable pH-millivolt meter.

3.2.5. Calculation of GHG emission and GHGI

The emission rate was calculated based on the equation from IAEA (1992) as follows:

$$E = \frac{Bm}{Vm} \; x \; \frac{\Delta C}{\Delta t} \; x \; \frac{V}{A} \; x \; \frac{273.2}{T + 273.2}$$

where E is the flux (mg m^{-2} min^{-1}), Bm/Vm (ρ) is the density of gas (mg m^{-3}), Δc/Δt is the average rate of concentration change with time (ppmv min^{-1}), V is the volume of the chamber (m^3), A is the base area of the chamber (m^2), and T is the temperature in the chamber (°C). The total CO_2, CH_4 and N_2O emissions were calculated according Singh *et al.* (1999):

$$CO_2, CH_4 \text{ or } N_2O \text{ emissions } = \sum_i^n (RixDi)$$

Where R_i is the rate of CO_2, CH_4 or N_2O flux (mg m^{-2} min^{-1}) in the i^{th} sampling interval, D_i is the number of days in the i^{th} sampling interval and n is the number of sampling intervals.

GHG emission equal with total CO_2 emission equivalent were calculated by using the following formula (IPCC 2007b):

GHG emission (kg CO_2 eq ha^{-1}) = (25 x CH_4) + (298 x N_2O)

Greenhouse gas intensity (GHGI) was calculated as described in Shang *et al*. (2010):

GHGI (ton CO_2 eq ton^{-1} grain yield) = GHG emission/yield

3.2.6. Statistical analysis

The effects of the treatments were analyzed with SAS software (SAS Institute 2003). The significant effects of different sites and steel slag applications were examined by using a two-way analysis of variance (ANOVA). While the effects of water managements were examined by using one-way ANOVA. When significant differences were detected at $P = 0.05$, the mean values were compared by using Tukey's HSD test (SAS Institute 2003).

3.3. Result and discussion

3.3.1. Methane flux

The CH_4 fluxes under different water managements are shown in Figure 3.3. The CH_4 fluxes of AWD and S-AWD were comparably low, there were quite huge amounts of CH_4 degassing from the AWD and S-AWD when the water was drained. The CH_4 fluxes from CF, AWD and S-AWD were approximately 205, 134 and 147 mg C m^{-2} day^{-1}, respectively. During the measurement period, the peak of CH_4 flux was in CF at 57 DAT (468 mg C m^{-2} day^{-1}). Total CH_4 emission during rice growing period from CF, AWD and S-AWD approximately ranged 219.4; 143.8 and 157.4 kg ha^{-1} $season^{-1}$, respectively. In this study, there were CH_4 reduction approximately around 34.5 and 28.4 % by application of AWD and S-AWD, respectively. Sass *et al.* (1992) reported that multi-aeration decreased CH_4 emission by 12% compared to continuous flooding, without any decreases in rice yield. Yagi *et al.* (1996) reported that CH_4 emission with intermittent irrigation decreased to 45% of continuous flooding. Katayanagi *et al.* (2012) reported that alternate wetting and drying has the potential to reduce CH_4 emission by 73 % compared with traditional flooded rice. In the paddy soil ecosystem, CH_4 is produced by microbial activities in the extremely anaerobic conditions that resulted from flooding soils by irrigation or rain water. A part of CH_4 produced in the anaerobic layer of soil is re-oxidized at the oxidized zones in soil, and the rest is transported to the atmosphere, mainly via plants. Water management practise have a strong influence on the processes involved in CH_4 emission on rice paddy fields. The presence of surface standing water is essential for the development of the anaerobic conditions paddy soil by limiting the transport of atmospheric oxygen into soil. Drying give to the higher Eh and to decrease the amount of CH_4 emitted.

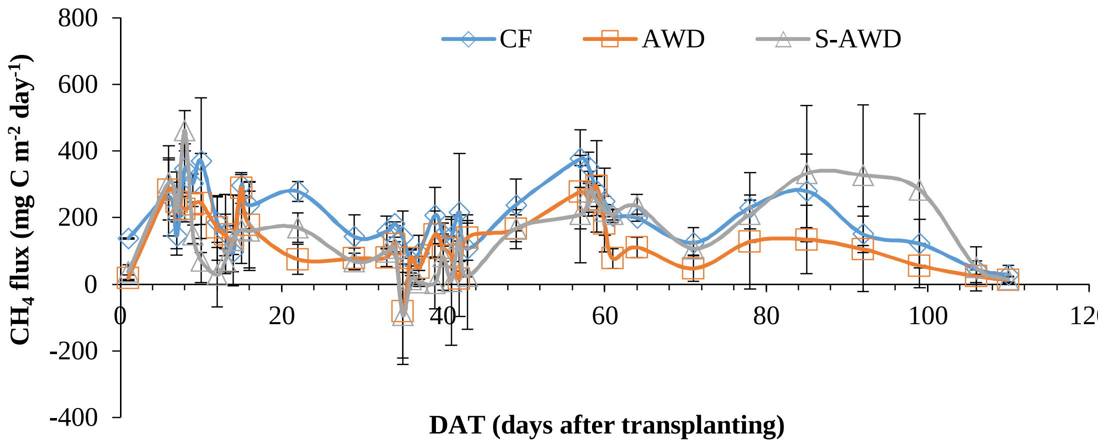


Figure 3.3. Dynamic changes of CH_4 fluxes under different water managements

3.3.2. Nitrous oxide emission

In general, the N_2O fluxes at different water managements were quite low. Statistics revealed that there is no difference in N_2O fluxes. N_2O fluxes from CF, AWD and S-AWD range approximately 234; 218 and 248 μg N m^{-2} day^{-1}, respectively. The three treatments showed pronounced emission peak at different time (Figure 3.4). As already mentioned in previous studies, continuous flooded paddy field have less significant N_2O emissions but emit great amounts of CH_4 (Smith and Patrick 1983; IPCC 1992, Neue and Sass 1998; Zou *et al.* 2005a; Yan *et al.* 2009; Cai *et al.* 1997), while there is a trend towards water-saving irrigation practices with less flooding which on the other hand reduce CH_4 emissions but increase N_2O emissions (Smith and Patrick 1983; Cai *et al.* 2001; Zou *et al.* 2005b; Johnson-Beebout *et al.* 2009; Liu *et al.* 2010; Peng *et al.* 2011). In this results were partly in contrast to the other findings as we found reduced CH_4 emissions under conditions of less flooding, indeed, but we also found less N_2O emissions with less flooding. AWD resulted lower N_2O emission, while S-AWD showed higher N_2O emission compare to CF. This result was similar to Berger *et al.*

(2013) that AWD could reduce CH_4 as well as N_2O emission from paddy field. The N_2O emission from CF, AWD and S-AWD were approximately 0.251; 0.233 and 0.265 kg ha^{-1} $season^{-1}$. As possible reasons for this finding, soil property such as the sandy substrate which most likely caused strong NO_3^- leaching, which may have made a huge N_2O production impossible. But this results also suggest that, with having N_2O production in the soil because short term environmental changes such as less flooding or short term fluctuations of the water table height during one vegetation period, may not have a strong effect on N_2O production in short term. While, N_2O is formed primarily from nitrification and denitrification in soil, depending on the aerobic and anaerobic conditions of soil (Mosier *et al.* 1998). Emissions of N_2O during intermittent irrigation periods strongly depended on the status of water logging in the fields. Different water regimes in rice fields caused a sensitive change in N_2O emissions (Zou *et al.* 2005). AWD give varies results in N_2O emission. Smith and Patrick (1983) observed that alternate anaerobic and aerobic cycling considerably increased N_2O emission relative to constant aerobic and anaerobic conditions. While Monteny *et al.* (2006) reported that drainage can suppress N_2O emissions by improving aeration.

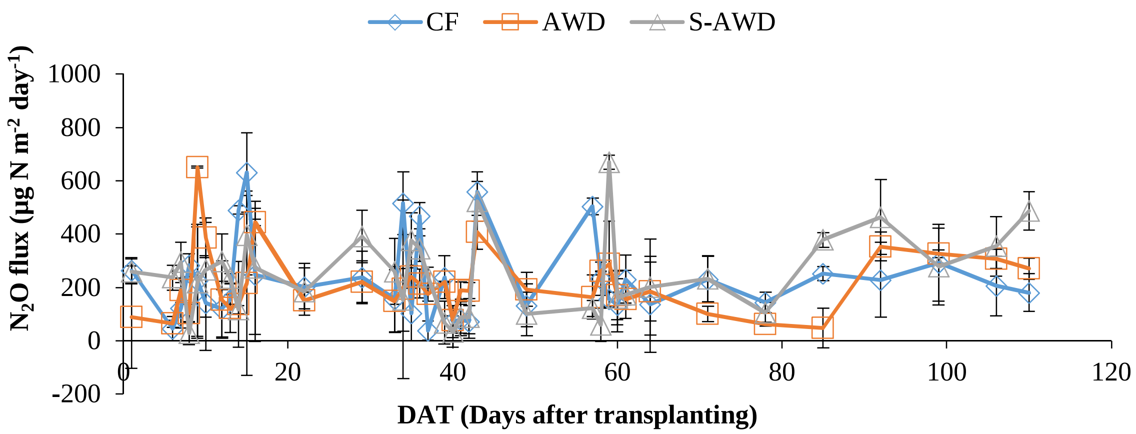


Figure 3.4. Dynamic changes of N_2O fluxes under different water managements

3.3.3. Soil pH and soil redox potential

There was no effect of different water management on soil pH. The pattern of soil pH is similar among to the treatments. The soil pH values on CF were around 4.78-6.54, while pH values from AWD and S-AWD ranged approximately 5.02-6.48 and 4.40-6.60, respectively (Figure 3.5a). Mostly, increasing soil pH near to neutral enhance CH_4 emission, i.e., the increase in the soil pH may have enhanced the activity of soil microorganisms, including that of methanogens, and this activity accelerated the decomposition of organic matter with an increase in CH_4 emissions. In this study, the reduction of CH_4 emission was not influenced by soil pH.

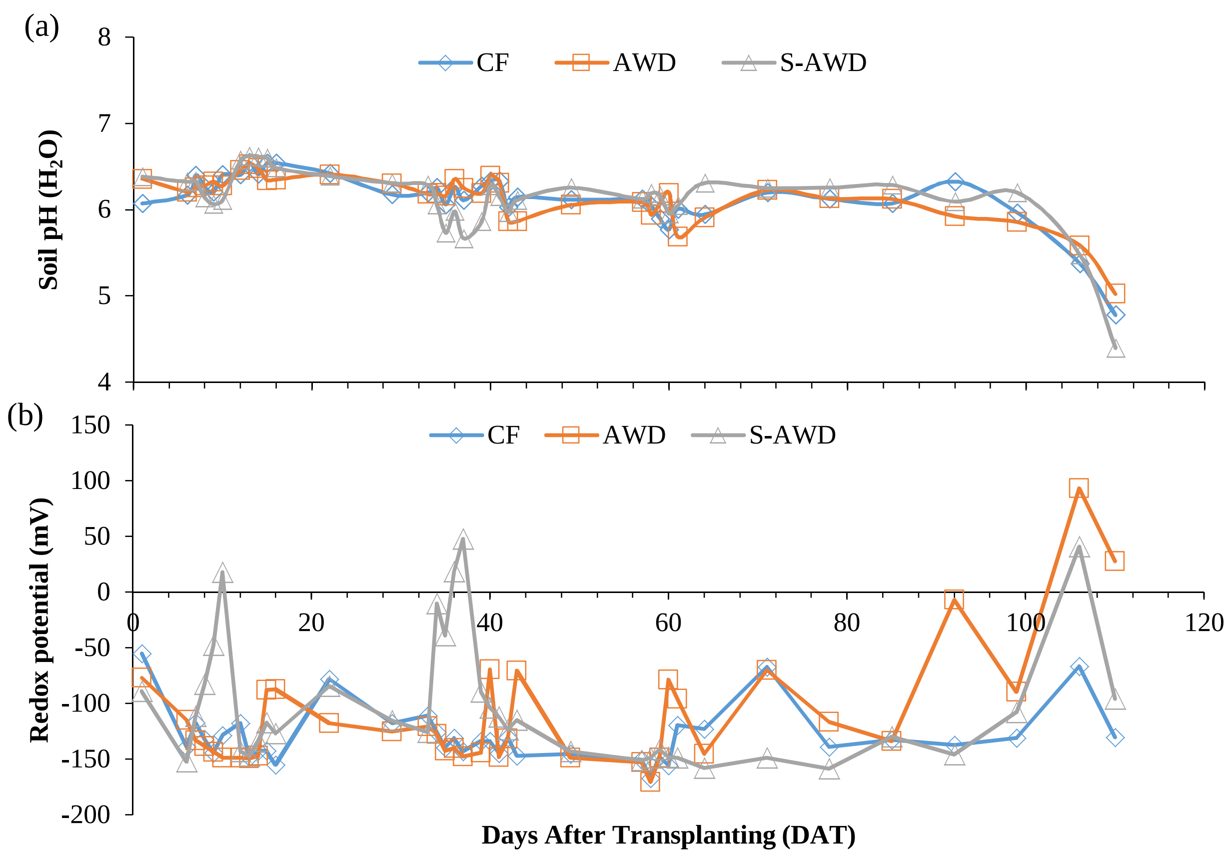


Figure 3.5. Dynamic changes of soil pH (a) and redox potential (b) under different water managements

Redox potential (Eh) pattern of the soil shows that water management influenced soil Eh. When the field was drained, soil Eh showed higher values (Figure 3.5b). Soil redox potential increased rapidly following the drainage of the field. AWD and S-AWD have higher values compare to CF. The range soil Eh values from CF were around -55 to 167 mV, while soil Eh from AWD and S-AWD were approximately -149 to 92 and -158 to 47 mV, respectively. Water managements influence GHG emission especially CH_4 by changing soil water content, which determines aerobic and anaerobic conditions in the soil. Aerobic and anaerobic conditions were related with soil Eh, which has been used as one of the most indicative soil parameters for CH_4 and N_2O from irrigated rice fields (Hou *et al*. 2000). Conrad (1999) identified three sequential phases in soil reduction dynamics: (i) H_2–dependent methanogenesis at positive Eh (360–510 mV), then (ii) sulfate or Fe (III) reductions when the first phase methanogenesis is becoming thermodynamically unfavourable, and (iii) vigorous acetate-dependent methanogenesis with a constant rate. The patterns in relation to soil Eh shows that CH_4 emission was highest when redox potential was lowest and N_2O emission was at a minimum at the same time (Hou *et al.* 2000). The critical soil redox potential for CH_4 and N_2O production has been demonstrated in laboratory studies to be below about -150 mV for CH_4 and above about -250 mV for N_2O (Wang *et al.* 1993; Masscheleyn *et al.* 1993).

3.3.4. Plant height and plant tiller

Water management practice showed similar response in vegetative and reproductive characteristics. No significant effect of AWD, S-AWD and CF were observed in plant height and plant tiller. Maximum plant tiller were 18 (Figure 3.6a), while plant height were 120 cm (Figure 3.6b). This condition most likely because the application of water management still give a sufficient amount of nutrient in root zone to secure a high photosynthetic rate (Osaki *et al.* 1997). Yang and Zhang (2010) that reported AWD improves water use efficiency and can improve yield by increasing the proportion of tillers that are productive, reducing the angle of the topmost leaves, (thus allowing more light to penetrate the canopy) and modifying shoot and root activity, implying altered root-to-shoot signalling of phyto-hormones such as abscisic acid (ABA) and cytokinins. While according to Koma and Sinv (2003) reported that reducing water in field lead to reducing adequate soil moisture that make less number of tiller per m^{-2}.

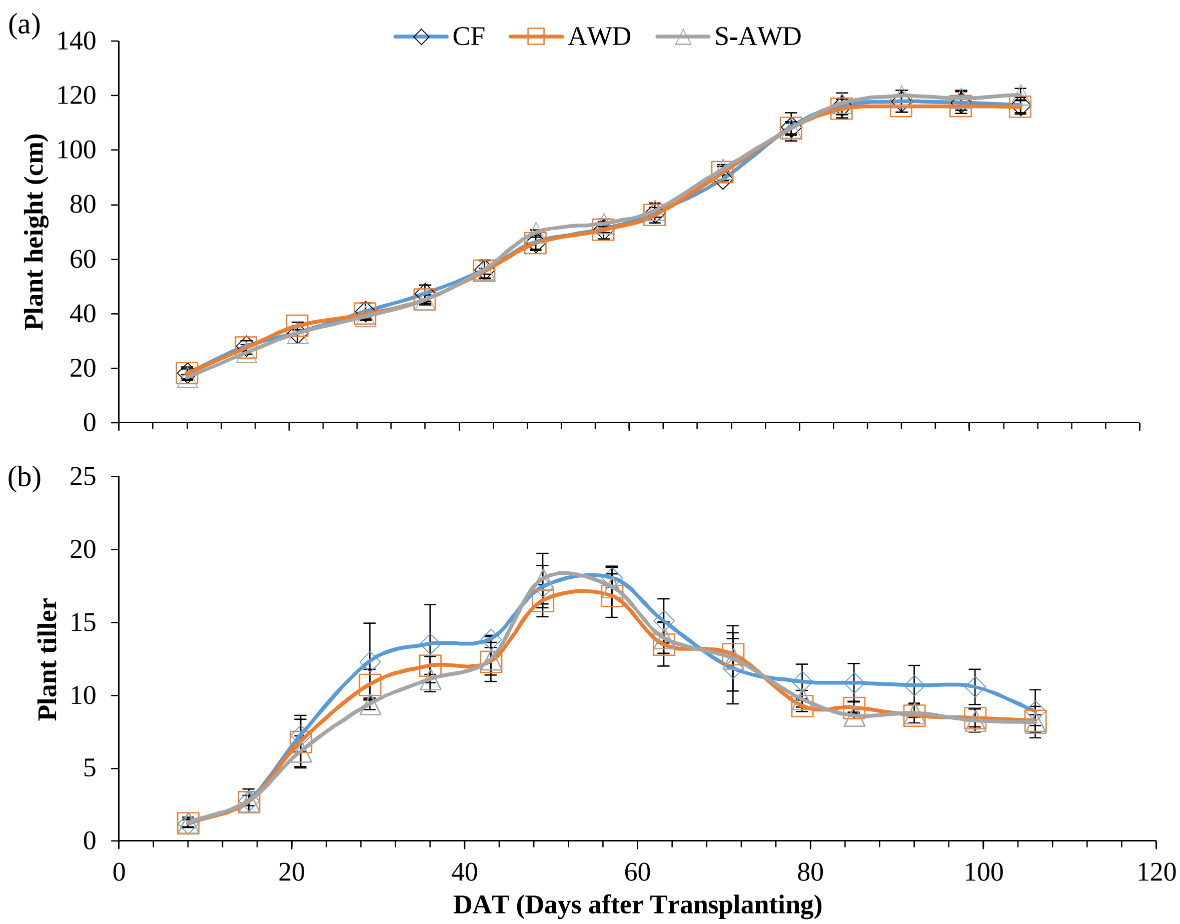


Figure 3.6. Plant height (a) and plant tiller (b) during rice growth under different water managements

3.3.5. Yield components

Yield components in all parameter measured between CF, AWD and S-AWD were not different. Among the yield components, no large different between the treatments that caused yield one treatment higher or lower than another. These included 1000 weight grain (30.5; 30.1 and 29.6 g), % filled grain (67.9; 67.4 and 63.2%), and harvest index (0.38; 0.38 and 0.35) (Table 3.3). The highest panicle number per m^2 was for CF (2408) and no significant differences with AWD and S-AWD. The total biomass ranged between 0.929-1.073 kg m^{-2}. No significant differences were observed among the water treatments on total biomass. Providing enough water for rice growth would promote percentage of filled grain as long as nutrient supply is sufficient and climate is favourable for rice plant growth (Yoshida 1981). In contrast with what was mentioned by Yoshida (1981), this study showed that water management do not affect the yield components. Tuong *et al.* (2009) stated that no yield penalty was observed when safe AWD was practiced. Bouman and Tuong (2001) summarized 31 field experiments on AWD and they found the yield reductions of 0–70% in AWD treatments compared with CF controls in 92% of the experiments. Hatta (1967); Tabbal *et al.* (1992), and Singh *et al.* (1996) reported that maintaining a very thin water layer, at saturated soil condition, or AWD can reduce water applied to the field by about 40–70 percent compared with CF, without a significant yield loss. Sato and Uphoff (2007) reported from Indonesian experience that CF was not essential for achieving high rice yields. The large variability in the performance of AWD was caused by differences in the irrigation interval, soil properties and hydrological conditions across the experiments. In addition, variety is a major factor that influences the performance of AWD (Peng and Bouman 2007). Contrary to Vizier (1990) and Sahrawat (2000), under CF condition rice yields tend to be very low due to detrimental to rice root growth, limited rice growth during the vegetative phase of rice and the chemical changes of paddy soil that affect the transformation of nutrient.

Table 3.3. Yield components of Cisadane under different water managements

Yield components	Treatments		
	CF	**AWD**	**S-AWD**
1000 weight grain (g)	30.5 ± 0.17	30.1 ± 0.71	29.6 ± 0.27
Unfilled grain	354 ± 91.9	354 ± 95.4	453 ± 91.9
Filled grain	761 ± 184.6	741 ± 161.1	803 ± 233.3
% filled grain	67.9 ± 6.6	67.4 ± 8.8	63.2 ± 7.4
Panicle number per m^2	2075 ± 736	1742 ± 245	2408 ± 1054
Panicle length (cm)	23.0 ± 0.40	23.5 ± 0.71	22.8 ± 0.62
Root dry weight (kg m^{-2})	0.116 ± 0.0294	0.115 ± 0.0361	0.116 ± 0.0376
Shoot dry weight (kg m^{-2})	0.844 ± 0.2345	0.814 ± 0.1216	0.957 ± 0.3077
Total biomass (kg m^{-2})	0.960 ± 0.2639	0.929 ± 0.1577	1.073 ± 0.3453
Grain harvest index	0.38 ± 0.084	0.38 ± 0.013	0.35 ± 0.062

3.3.6. Nutrient uptake

Table 3.4 shows nutrient uptake by Cisadane under different water managements. Water treatments did not significantly affect the uptake of N, P, K, Ca, Mg, Fe, Mn, Zn and Si by rice plants during their growth. According to Yang *et al.* (2004) that AWD methods enhance nutrient uptake because it can improve root morphology and root activity (Yang *et al.* 2004). Intermittent irrigation is believed to improve oxygen supply to rice root system with potential advantages for nutrient uptake (Stoop *et al.* 2002). Bonkowski (2004) has indicated that under more aerobic soil conditions, there will be larger populations of soil fauna that contribute to biological processes for supplying N needs of plants. Paddy soils characterized by high amount of Fe- and Mn-oxides and low cation exchange capacity, aerobic condition reduce the accumulation of soluble ferrous iron and manganese after submerging, which are toxic to rice plants under the continuously flooded conditions (Olaleye *et al.* 2001). However, contrary to their studies, Levit (1980) reported that nutrient uptake by crop plants is generally decreased under water-stress conditions owing to a substantial decrease in transpiration rates and impaired active transport and membrane permeability and resulting in a reduced root-absorbing power of crop plants nutrient uptake from the soil solution is also closely linked to the plant root and soil water status. A decline in the soil moisture content is associated with a decrease in the diffusion rate of nutrients from the soil matrix to the absorbing root surface. Rice need silicon (Si) in large amounts for vigorous growth and high production. Silicon deposited in the leaves, stem and husk. The function of Si in plant are to mitigate fungal infection and pest attack, alleviates lodging and other abiotic stress, improves the light-interception ability by plants in a community and minimizes transpiration losses (Ma and Takahashi 2002).

Table 3.4. Nutrient uptake by Cisadane under different water managements

Water managements	N	P	K	Fe	Mn	Zn	Si
	--------g kg^{-1}--------			-------mg kg^{-1}--------			g kg^{-1}
CF	19	7	29	39	205	10	33
AWD	14	9	20	38	149	13	32
S-AWD	13	7	23	39	153	8	33

.3.7. Water saving and water productivity index (WPI)

The CF treatment needed supply water approximately around 765.7 m^3 ha^{-1} followed by AWD and S-AWD ranged approximately 712.4 and 677.0 m^3 ha^{-1}, respectively (Table 3.5). Although no significance effect of water treatment on water supply in paddy field, AWD and S-AWD could save the water approximately 6.96 and 11.59%, respectively. According to Bhuiyan (1992), under traditional practices in the Asian tropics and subtropics rice requires water between 700-1500 mm per cropping season depending on soil texture. Tuong and Bouman (2003) reported that the total water input varies from 700 to 5300 mm for 100 day per season in lowland rice field in tropic, depending on climate, soil characteristics and hydrological conditions. Although no significance different, S-AWD and AWD resulted higher WPI compare than CF. Study from Chapagain *et al.* (2011) showed that total water required in AWD plot less 29% as compare to conventionally flooded plot in Chiba Japan, while WPI was significantly higher (1.7 kg m^{-3}) than conventional irrigation (1.3 kg m^{-3}). Bhuiyan and Tuong (1995) reported that a standing depth of water throughout the season is not needed for high rice yields. They added that about 40–45 percent of the water normally used in irrigating the rice crop in the dry season was saved by applying water in small quantities only to keep the soil saturated throughout the growing season, without sacrificing rice yields. Alternate drying and wetting of the fields allows for good aeration of the soil and better root growth thereby increasing rice yield and water use efficiency in Indonesia (Uphoff, 2006).

Table 3.5. Water supply, water saving and water productivity index (WPI) under different water managements

Treatments	Water supply (m3ha-1)	Water saving (%)	WPI (kg m-3)
CF	765.7		7.44
AWD	712.4	6.96	8.10
S-AWD	677.0	11.59	8.27

3.3.8. Total GHG emission and GHGI

The effects of water managements on the GHG emission, rice yield, and GHGI are presented in Figure 3.7. GHG emissions from CF, AWD and S-AWD were 6.97, 4.63 and 5.08 ton CO_2 eq ha^{-1} $season^{-1}$, respectively. Although, there was no statistical difference, there were tendency of GHG emission reductions approximately 33.6 and 27.2% due to application of AWD and S-AWD, respectively. Hadi *et al.* (2010) and Feng *et al.* (2013) reported 34 and 54% less GWP (CH_4 and N_2O) of intermittent irrigation as compared with traditional flooding. The AWD and S-AWD did better than the CF treatment in term of GHG emission. Therefore, it can be expected that AWD and S-AWD would usually produce less GHG emission than CF. To evaluate the climate implication of the cultivation practices, it is desirable to have relative contribution of each gas to global warming. In this study, CH_4 emission from different water management account for 95-97% of the contribution to global warming, while N_2O emission is only contributed 3.4-4.9%. No significance difference on yield but calculated yields were highest under AWD which produced better than under CF conditions. Yield of AWD was 1.3% higher than under CF conditions. It has been reported by Tuong *et al.* 2005; Yang *et al.* 2007; Zhang *et al.* 2008 that AWD can maintain or even increase grain yield because of the enhancement in root growth, grain-filling rate, and remobilization of carbon reserves from vegetative tissues to grains, when it compared with CF conditions. The lower value of the GHGI from AWD and S-AWD compared to control means that the treatments give more advantages to mitigate GHG emission and produce more rice.

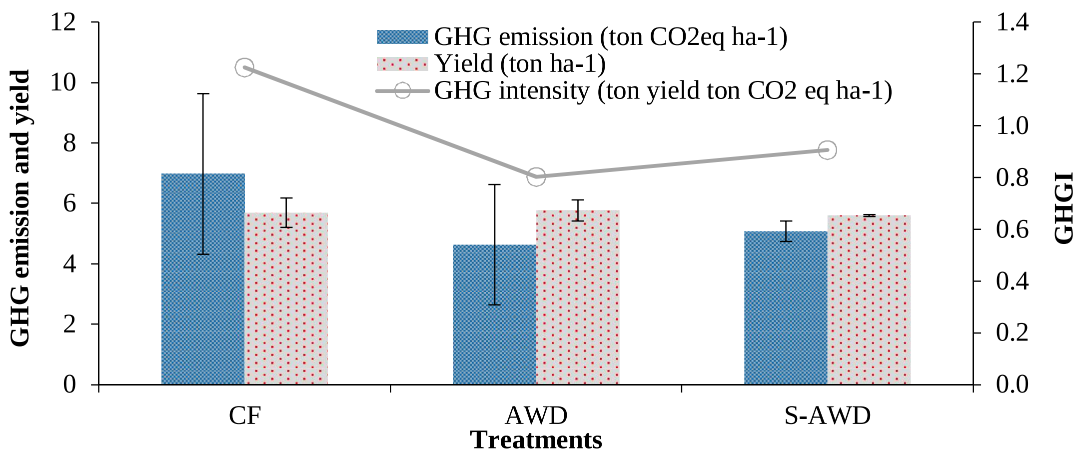


Figure 3.7. Effects of water managements on GHG emissions, yield and GHG intensity at paddy field during rice growing season

3.3.9. Relationship between CH_4-N_2O production and CH_4-N_2O emission from Pati's soil

Regression analysis was done between CH_4-N_2O production from chapter 2 as *y* axis and CH_4-N_2O emission from this chapter as *x*. Pati's soil was used in this regression analysis. The relationship between CH_4-N_2O production and CH_4-N_2O emission is shown in Figure 3.8. Results show that there were significant relationship ($P \leq 0.01$) between potential CH_4-N_2O production and CH_4-N_2O emission. This results can be used to predict CH_4-N_2O emission from rice field using CH_4-N_2O production from the same soil under incubation experiment. The equation from linear regression for CH_4 is $y = 44.189x$, with coefficient determination (r) of 0.46, while for N_2O is $y = 337.91x$, r = 0.47. Thus, based on the equation, CH_4 emission equals to 0.023 (2.3%) of the potential CH_4 production and, N_2O emission equals to 0.003 (0.3%) of the potential N_2O production. The CH_4-N_2O production resulted higher value than CH_4-N_2O emission from rice field.

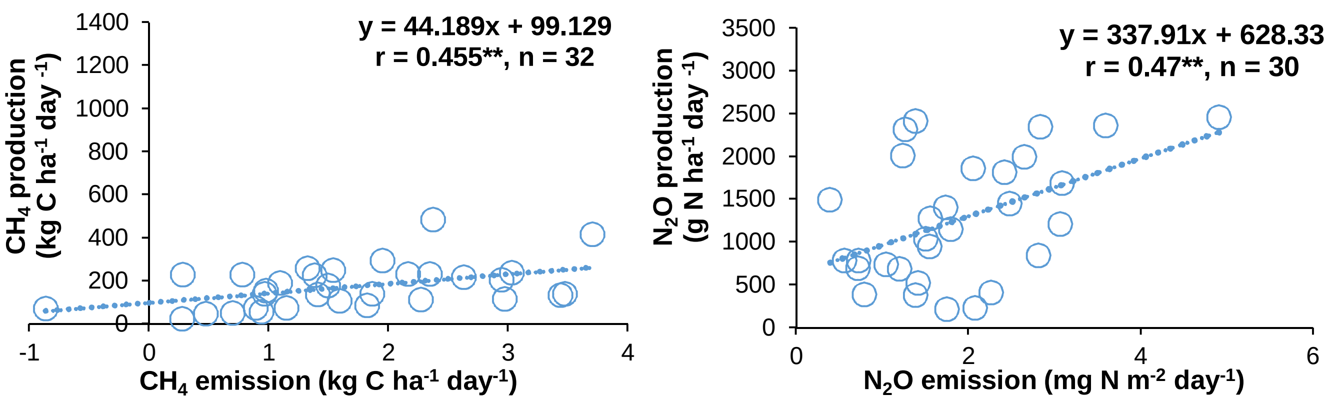


Figure 3.8. Relationship between potential CH_4 – N_2O production (*y*) and measured CH_4 – N_2O emission (*x*) from rice soil in Pati, Central Java, Indonesia

3.4. Conclusions

Field experiments on water management method of cultivating rice have demonstrated the utility of AWD and S-AWD for water saving in irrigated rice farming. This experiment also indicated that Water Productivity Index increased from continuous flooded irrigation. This field experiment confirms that AWD and S-AWD is a promising method in irrigated rice cultivation with benefits on water saving and maintaining the productivity comparable to continuous flooded irrigation. The increased productivity of water and its resource saving aspects are likely to be the critical factors that will make farmers and other stakeholders adopt AWD in water-scarce areas. However, it is difficult to draw general conclusions as AWD and S-AWD methods adopted in a certain area may not transfer to other areas because of variability in topography, soil, and climatic conditions across the rice agro-ecological domains. Therefore, it is important that comparative studies be conducted in different environments to verify this practice as a way to conserve water under conditions of water scarcity while maintaining, or increasing, crop yields. Moreover, long-term experiments are required to predict water management impacts on soil organic matter and provide leading indicators of sustainability, which can serve as an early warning system to detect impairments that threaten future productivity

CHAPTER 4

INFLUENCE OF WATER TABLES AND SOIL AMELIORATIONS ON GREENHOUSE GAS EMISSIONS FROM INDONESIAN PEAT SOIL COLOUM

4.1. Introduction

In our previous chapter, it has been observed that of AWD and S-AWD resulted lower GHGI as well as higher WPI compare to CF irrigation. This field experiment confirms that AWD and S-AWD is a promising method in irrigated rice cultivation with benefits on reduce GHG emission, water saving and maintaining the productivity comparable to continuous flooded irrigation. However, the area of rice cultivation in irrigated area is expected to reduce due to the land conversion. The option to fulfil the increasing food demand is looking toward the areas of new arable land including peatland. However, utilization of natural peatland cause changes on ecosystem. Therefore, utilization of peatland for agriculture is better conducted in degraded peatland, i.e., ex-burned peatland because peat fire is a major cause of peatland degradation that leads to loss of biodiversity and carbon stocks. Sustainable agriculture means increase the production as well as ecology adaptive to environment. Thus, the experiment in this chapter used degraded peat soil to examine water table and soil ameliorant on GHG emission for future agriculture usage in peatland.

Large areas of tropical forest peatland in Indonesia have been converted into agricultural and non-agricultural sectors because of human population growth and economic development. Approximately 14.9 million ha of peatlands are found in Indonesia, which are estimated to account for 47% of the total tropical peatland area (Ritung *et al*. 2011; Page *et al.* 2011). Peatland has huge amount of carbon stock and nitrogen which could be a source of of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (Inubushi *et al.* 2003). To be used for agricultural activities, peat soils need to be drained, limed and fertilized due to excess water, low nutrient content and high acidity. Regarding agricultural practices in wetland including peatland in Indonesia, there is indigenous knowledge of organic matter management that combines minimum tillage known as *tajak*, *puntal*, *hambur* systems, followed by application of ash, salt, and manure (Noor 2012). Therefore, water adjustment and soil amelioration are needed for suitable cultivation in peatland area due to the excess water and low soil fertility. However, lowering the water table increases carbon mineralization and CO_2 and N_2O emissions but decreases CH_4 emission (Moore and Dalva 1993; Regina *et al.* 1996; Berglund and Berglund 2011). Soil ameliorants are applied not only to enhance the nutrient status of the soil and to improve crop yield but also to reduce GHG emission. Most of the studies on soil ameliorations/amendments have been conducted in mineral soil; however, there are few publications on the effects of soil amelioration in peat soil.

Most studies in tropical peatland were based on remote sensing data (Jaenicke *et al.* 2008). In the field, daily GHG emissions vary depending on climate condition and hydrologic regime. Therefore, we conducted this study by investigating the influence of water depth and soil amelioration on peat soil columns adjusted to the same conditions. The objective of this chapter is to discuss the effect of water table and soil amelioration on GHG emissions from Indonesian peat soil columns.

4.2. Materials and methods

4.2.1 Site and soil sampling

A soil sampling site was selected in Jabiren, Pulang Pisau district, Central Kalimantan, Indonesia (S 02^{0}30’52.5”; E 114^{0}10’11.6”). The soil sampling area is bounded by Sungai (river) Jabiren and Sungai Kahayan. The water level of this area is controlled by small canals that are approximately 50 cm deep and 50-75 cm wide, respectively. The canals were constructed from 1995 as part of the Mega Rice Project (Firmansyah *et al.* 2013). The water table at this site is approximately 30-60 cm below the soil surface. The peatlands in this area are dome shaped (Hikmatullah *et al.* 2013). This area was burned in 2005. The peatland areas are extensive and there are various types of land-use managements; however the soil was sampled in a fallow area under natural vegetation and near rubber plantations which were established in 2006. The soil sampling location was covered with fern and grasses.

Table 4.1. Chemical characteristic of peat soil collected from Central Kalimantan, Indonesia

Parameters	Values
pH	
H_2O	3.1
KCl	2.3
EC (dS m^{-1})	0.028
Organic (g kg^{-1})	
C	336.5
N	9.7
C/N	35
Available P (mg kg^{-1})	3.49
Available K (mg kg^{-1})	22.41
Exchangeable cations ($cmol_c$ kg^{-1})	
Ca	20.37
Mg	2.07
K	0.05
Na	0.17
CEC	88.45
Base saturation (%)	26
Ash content (g kg^{-1})	187.9
Silicate (g kg^{-1})	45.2
Humic acid (%)	10.91
Pyrite (%)	0.06

Twenty-seven peat in column (0-100 cm) samples were collected in March 2014. The peat samples were collected approximately 20 m from the small canal. The peat depth at this site was approximately 5-7 m (Firmansyah *et al.* 2013). The decomposition status of the peat material is dominantly classified as hemic (moderately humified) and fibric (less humified) but in the surface were classified as sapric (most humified) (Hikmatullah *et al.* 2013). The characteristics of the soil properties are listed in Table 4.1. To collect the peat samples, the top of the peat soil was removed using knife approximately 2-3 cm. The soil columns were collected using with polyvinyl chloride (PVC) pipe with a diameter of 21 cm and a length of 100 cm. The PVC pipe was inserted vertically to the soil to a depth of 100 cm and carefully pulled up with the soil inside after sealing the bottom. The litter inside the soil columns was not removed. The base of the soil columns was closed permanently with a tight cap during excavation, while the top of the columns was sealed with removable cap after the excavation. The columns were transported to Indonesian Agricultural Environment Research Institute (IAERI), Jakenan, Central Java, Indonesia.

4.2.2. Experimental design

After been transported to IAERI, the excess water from the soil columns was allowed to drain to change the air-filled porosity of the peat samples. The peat samples were weighed to determine the moisture content. Then, holes were made in each soil column based on the water depth position. Each soil column was dipped into large bucket in the greenhouse to control the water depth. The water depth of each bucket was checked using the transparent tube (diameter of 10 mm) that was installed on the outside wall of each the bucket. The water depths were checked daily to ensure they remained at constant level. The columns for the water depth manipulation are shown in Figure 4.1. Rain water was used to set the water depths. A randomized block design with two factors was used to establish 3 different water depths (15 cm, 35 cm and 55 cm from the soil surface) and 3 different ameliorants (without ameliorant/control, 2.5 Mg ha^{-1} biochar + 2.5 Mg ha^{-1} compost; 2.5 Mg ha^{-1} steel slag + 2.5 Mg

ha^{-1} compost), replicated 3 times. According to Susilawati *et al.* (2015) that steel slag applications at 1 Mg ha^{-1} and 2 Mg ha^{-1} could decrease the CH_4 and N_2O emissions from paddy rice field but the effect was not statistically significant, thus the rate of steel slag application was increased in this study. The experimental design in this study was adopted from Aerts and Ludwig (1997); Funk *et al.* (1994), and Jungkunst *et al.* (2008); these studies aimed to maintain water in peat soil columns at depths of 0-10 cm; 5-30 cm and 5-40 cm below soil surface for experimental periods of approximately 60, 130 and 141 days, respectively. The water depth treatments in the soil columns were initiated 11 days before the first gas sampling. Data were collected from April-July 2014.

4.2.3. Biochar, steel slag and compost preparation and application

The biochar used in this experiment was produced by pyrolysis of empty fruit bunches of oil palm. Specifically, the empty fruit bunches of oil palm were placed in pyrolysis reactor and burned at a temperature of approximately 250-300^{0}C for 8 hours. Typically, a yield of 20-30% of biochar was achieved. The biochar was then ground to pass through a 2 mm stainless steel sieve and mixed thoroughly to obtain a fine granular consistency. The steel slag that was used in this experiment is a by-product of the steel industry and was obtained from West Java, Indonesia. The steel slag was ground and passed through a 2 mm sieve before use. The compost was a mixture of cow manure, filter cake of sugarcane, miller's bran and lime. The heap of organic materials was maintained under minimum oxygen conditions for 1 month. The heap was watered and mixed thoroughly twice a month. The chemical properties of the ameliorants are shown in Table 4.2. The ameliorants were thoroughly mixed with the water, and these mixtures were applied to the soil surface 1 day before the first gas sampling. The water and ameliorant mixture was used to ensure that the ameliorants were more effectively absorbed at and below soil surface.

Table 4.2. Chemical properties of ameliorants used in this experiment

Parameters	Compost	Steel slag	Biochar
pH (H_2O)	7.66	8.24	9.96
Organic C (g kg^{-1})	240.8	3.9	468.2
Total N (g kg^{-1})	18.0	0.3	19.0
Chemical composition (%)			
P_2O_5	4.38	0.27	1.05
K_2O	0.29	0.07	0.22
Al_2O_3	0.96	2.00	0.70
SiO_2	13.5	29.2	6.00
MnO	0.65	4.52	0.39

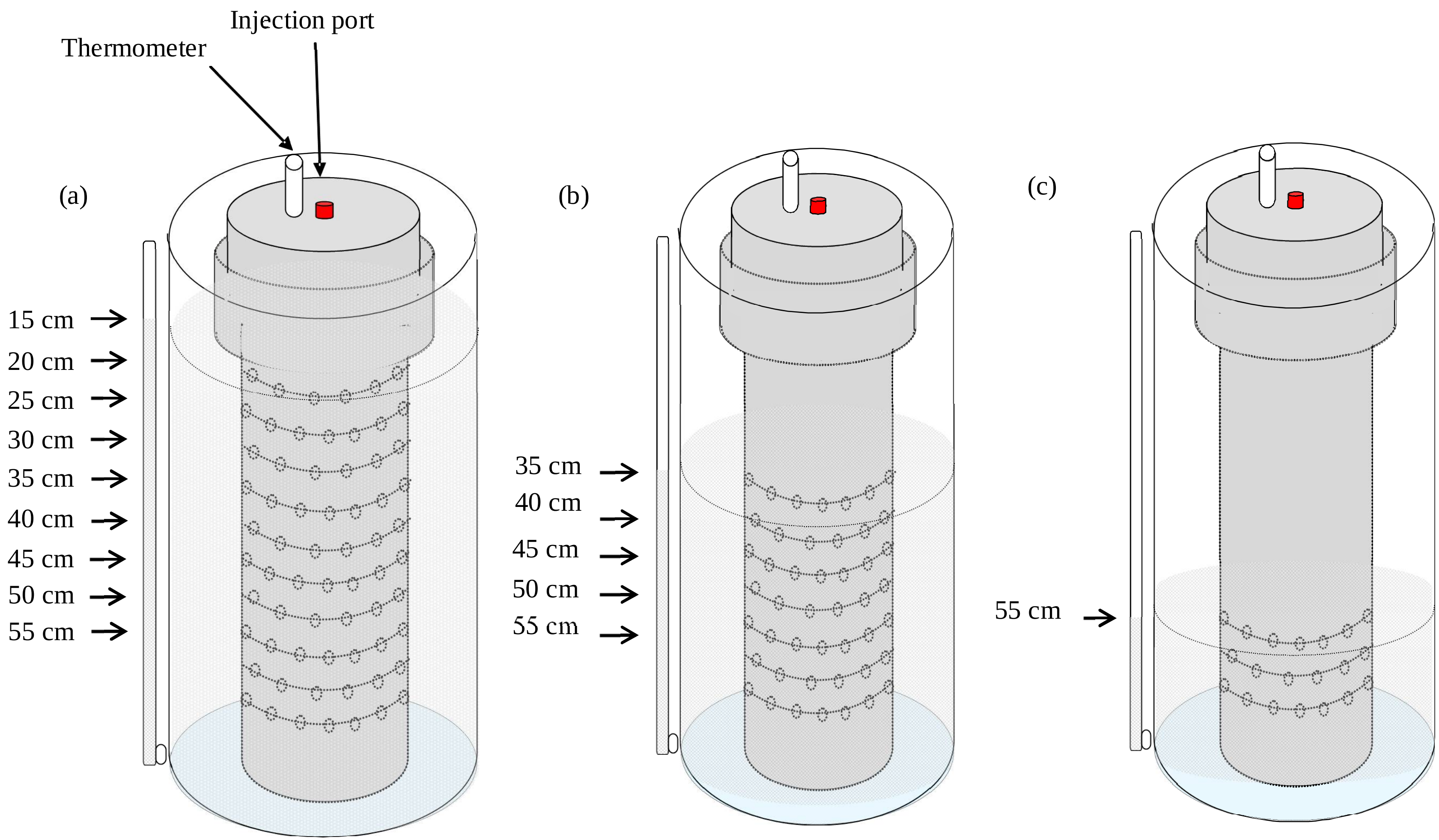


Figure 4.1. Sketch of the equipment used for water table treatments: 15 cm (a), 35 cm (b) and 55 cm(c).

4.2.4. Gas measurement

Gas samples for CO_2, CH_4 and N_2O from the soil columns were measured simultaneously once a week. Gas samples were measured by a closed dark chamber method. The caps of soil column were used as the chambers. The height of the chamber was 20 cm. Each of the caps of the soil column was equipped with rubber septum for taking gas sampling and also equipped with thermometer for measuring temperature inside the chamber. Gas samples from the inner chamber were taken once a week using 10-cm^3 syringes and repeated the sampling 5 times every 5 minute (5, 10, 15, 20 and 25 minutes). Gas samplings were started from 06:00 in the morning on each sampling day. The caps were removed after gas sampling to allow the peat surface to be in contact with ambient air. Gas sample in the syringes were transferred directly to the laboratory and were determined by gas chromatography (GC). A GC is equipped with a thermal conductivity detector (TCD) for CO_2 analysis, a flame ionization detector (FID) for CH_4 analysis and an electron capture detector (ECD) for N_2O analysis. The calculation of CH_4 or N_2O fluxes were described already in previous chapter.

Other parameters were measured at the same time with gas sampling. The parameters were soil pH, Eh and soil-water temperature. Soil temperatures were measured, using a digital thermocouple. In each soil column, the redox electrode was permanently installed at a depth of about 5 cm below the peat surface. Redox potential and soil pH were measured weekly using a portable pH-millivolt meter.

4.2.5. Statistical analysis

A two-way analysis of variance (water depths and ameliorations) followed Tukey's honestly significant difference (HSD) test was used to compare the mean values of CO_2, CH_4 and N_2O fluxes. The relationship between gas fluxes and each of the treatments, the treatments and the soil parameters were done by simple regression. Statistical considerations were based on $P < 0.05$ and $P < 0.001$ significance levels. Statistical analyses were conducted using SAS 9.1.3 portable (SAS Institute 2003).

4.3. Results and discussion

4.3.1. Carbon dioxide emissions

The CO_2 fluxes patterns for the various treatments are shown in Figure 4.2. During the first week of the experiment, the CO_2 fluxes from the peat soils were high for all treatments except without ameliorant in different water depths. These responses likely occurred because of the effect of ameliorants on soil pH. High soil pH measured early on the experiment (Figure 4.11). From that time onwards, there was no difference in CO_2 fluxes between all the treatments until 57 days after ameliorations (DAA) and the ranged from 13 to 275 mg C m^{-2} $hour^{-1}$. The CO_2 fluxes from the biochar+compost treatment at water depths of 35 cm and 55 cm exhibited high peaks at 64 and 71 days after amelioration (DAA), approximately 400 and 344 mg C m^{-2} $hour^{-1}$, respectively, probably because the decomposition of organic matter was more rapid at lower water depth. After the high peaks and then declined at the end of the experiment. In this study, the mean daily CO_2 emissions ranged from approximately 0.80-2.68 g C m^{-2} day^{-1}. Moore and Dalva (1993) measured CO_2 emissions of approximately 0.17-3.80 g C m^{-2} day^{-1} in peat soil columns from subarctic fen, temperate bog and temperate swamp at temperatures of 10 and 22.6^{0}C. According to Funk *et al.*

(1994), the CO_2 emissions from different water depth of taiga bog microcosms were approximately 0.8-3 g C m^{-2} day $^{-1}$. The Moore and Dalva (1993) and Funk *et al.* (1994) studies were also conducted using columns and the subsurface of the peat soil was used. CO_2 emissions from tropical peat soil are higher than those from temperate and boreal peat soil due to the temperature and moisture content, which influence the microbial processes leading to the production of these gases (Berglund *et al.* 2010). Based on a field study conducted in Jambi, Indonesia, Furukawa *et al.* (2005) reported that CO_2 emissions from tropical peatlands under different land-use management (coconut field, pineapple field and swamp forest) ranged from approximately around 0.72-6.38 g C m^{-2} day $^{-1}$. In addition, field studies conducted in secondary forest and paddy field in tropical peatlands, South Kalimantan resulted in CO_2 emissions of approximately around 3.36-6.62 g C m^{-2} day $^{-1}$ (Hadi *et al.* 2005). The higher CO_2 emissions recorded in field studies compared with the soil columns likely occur because peat soil in the field have different substrates and microbial populations, which develop in response to long-term differences in water table position and soil managements (Moore and Dalva 1993).

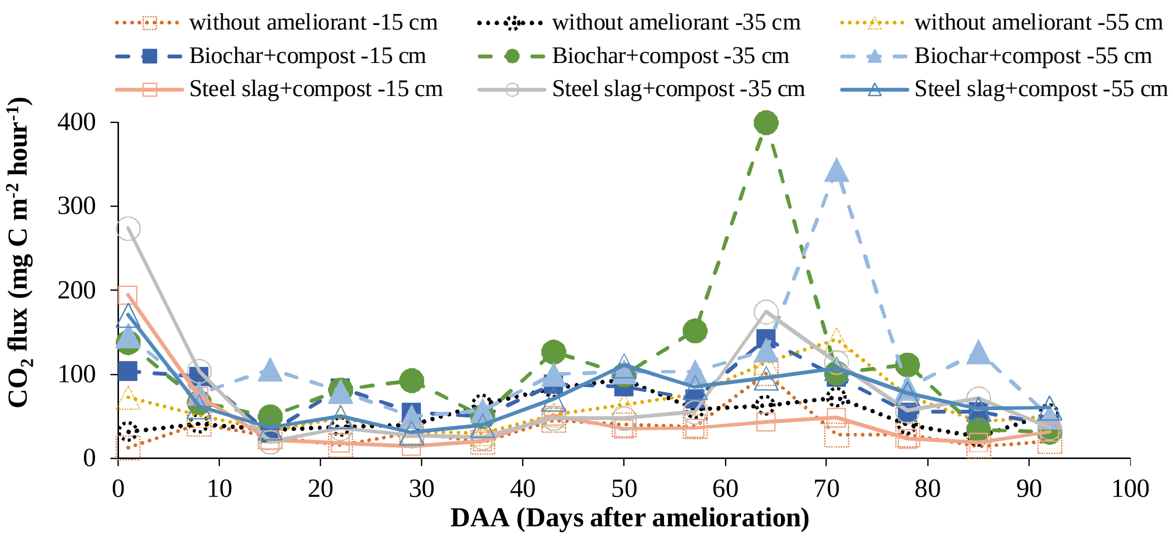


Figure 4.2. The changes in CO_2 fluxes from 3 different water depths and ameliorations during the experimental period.

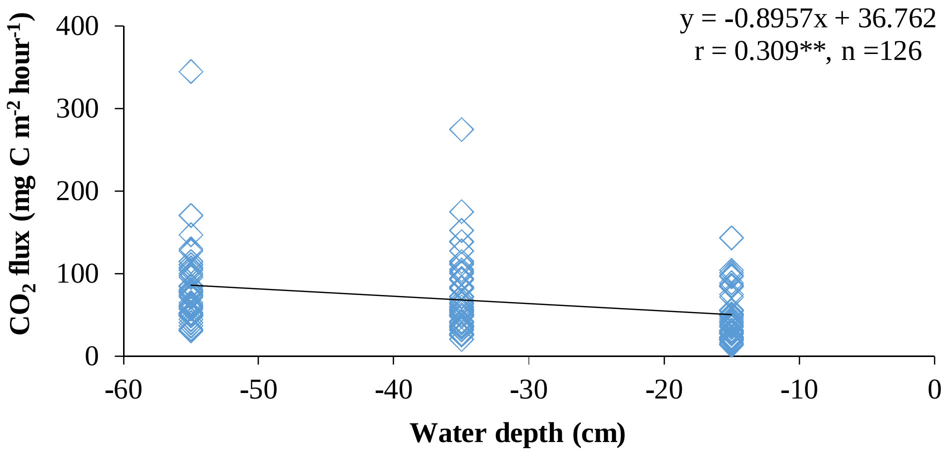


Figure 4.3. Relationship between water depths and CO_2, fluxes in peat soil (** indicates $P < 0.01$)

Figure 4.3 shows that there was relationship between CO_2 fluxes and the water depths in the three sets of columns (r = 0.309, n = 126, $P < 0.01$). This result is similar to the finding reported by Funk *et al.* (1994); Moore and Knowles (1988) and Moore and Dalva (1993), i.e., that CO_2 emission from peat soil are related to water depth. In this study, the CO_2 emissions from peat soil at lowering the water depths (35 and 55 cm) were approximately 56 and 62% higher, respectively, than 15 cm water depth. According to Jungkunst *et al.* (2008), lowering of water depth by 20 and 40 cm from soil surface increased CO_2 emission from temperate forest (in columns) by 33 and 65%, respectively. This likely occurred because oxygen diffusion into soil increases when water depth is lowered; therefore soil become aerated, allowing aerobic decomposition (Silvola *et al.* 1996; Nykanen *et al.* 1998). Consequently, CO_2 fluxes from soils increase under aerobic conditions (Moore and Dalva 1993). This condition increased the soil Eh, and this condition is known to favor microbial activity and nitrogen mineralization (Ueda *et al.* 2000). However, Aerts and Ludwig (1997) found that CO_2 emission from mesotrophic peat with high water-table was higher than from peat with lower water-table probably because the peat layers were not completely decomposed. Studies conducted by Lafleur *et al*. (2005) and Nieveen *et al.* (2005) indicate that the correlation between water depth and CO_2 emission is poor in temperate low shrub peatland.

The annual CO_2 emission from without ameliorant was approximately 4.4 ton C ha^{-1} year $^{-1}$, while the annual CO_2 emissions from steel slag+compost and biochar+compost treatments were approximately 5.8 and 8.7 ton C ha^{-1} year $^{-1}$, respectively (Figure 4.4). The CO_2 emission from control, slag+compost and biochar+compost treatments were approximately 50.2; 66.3 and 99.3 mg C m^{-2} hour^{-1}, respectively. Amelioration had a highly significant ($P < 0.01$) effect on CO_2 emissions which were enhanced by the application of biochar+compost and steel slag+compost. Steel slag+compost and biochar+compost application to the peat soil stimulated CO_2 emissions by approximately 1.4 and 4.3 ton C ha^{-1} year $^{-1}$, respectively. The CO_2 emitted from the biochar+compost treatment was nearly 2 fold higher than from without ameliorant likely because of an increased availability of the media as microbial substrates and an increased microbial decomposition and mineralization of organic matter (Smith *et al.* 2010; Jones *et al.* 2011). Although CO_2 emission from biochar+compost and steel slag+compost were quite higher than without ameliorant but the emissions were slightly lower compare the studies in peat soil from ICCTF (2011) and Sakata *et al.* (2015) (Table 1.1). The application of steel biochar+compost and slag+compost increased CO_2 emission. This result contradict with the study from ICCTF (2011), application of biochar from rice husk could reduce CO_2 emission in peat soil because organic matter as an ameliorant in peat soil requires one to consider the quality and type of materials and level of maturity of the organic matter. According to Ali *et al.* (2008), the effect of steel slag fertilizer on CO_2 production exhibited an increasing trend, which is indirect evidence of methanotrophic' activity. Another reason for the higher CO_2 emissions in response to amelioration is the soil pH. The biochar+compost and slag+compost treatments increased the soil pH. This finding indicates that there is a decreasing CO_2 emissions trend for water depths closer to the peat surface and increasing CO_2 emissions trend when ameliorations are applied.

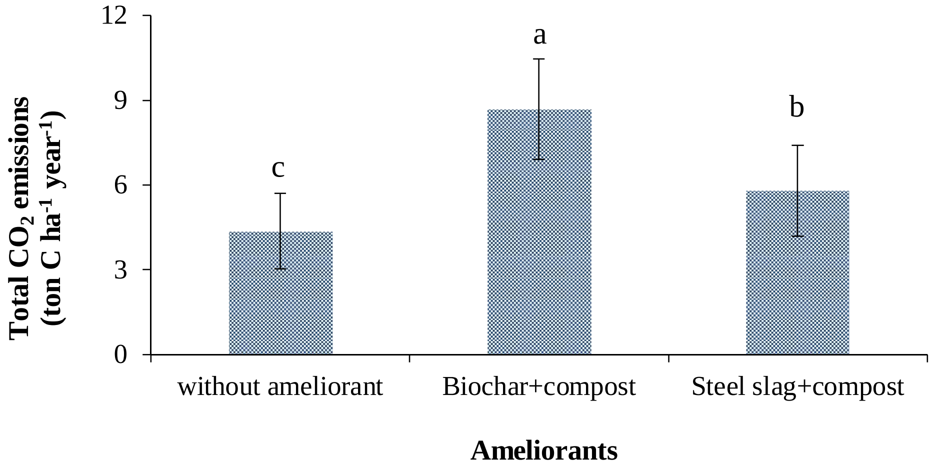


Figure 4.4. The CO_2 emission from different ameliorants at peat soil columns ($P < 0.05$)

4.3.2. Methane emissions

In this study, the CH_4 fluxes were very sporadic and there was no pattern between the treatments was observed throughout the measurements periods (Figure 4.5). The coefficients of variation within treatments were higher than 45% and sometimes exceeded 100%. Consequently, CH_4 emissions were not significantly affected by the treatments. A very high variation in the CH_4 fluxes were observed between the treatments during the first 29 days of measurements. The highest CH_4 fluxes, i.e., approximately around 0.307 mg C m^{-2} hour^{-1}, were recorded in the biochar+compost treatment at a water depth of 35 cm after which a sharp decrease was observed. The CH_4 fluxes during the 92 days of measurements were rather low and sometimes negative. Negative values indicate net uptake from the atmosphere by the ecosystem.

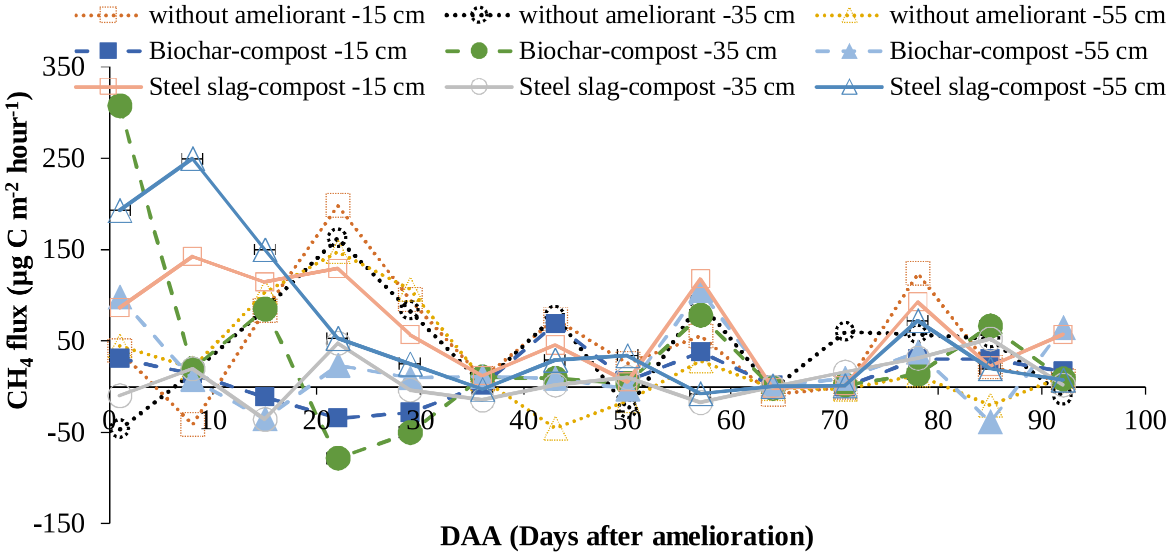


Figure 4.5. The changes in CH_4 fluxes from 3 different water depths and ameliorations during the experimental period.

The mean daily CH_4 emissions from all treatments are ranged from 0.17-1.51 mg C m^{-2} day^{-1}. The CH_4 emissions in this study were very low compared with similar experiments using peat soil columns, e.g., Moore and Dalva (1993). According this authors, the mean CH_4 emissions from subarctic fen, temperate bog and temperate swamp (using soil columns) ranged from 0.53-97.19 mg C m^{-2} day^{-1} and there was no significant relationship between water depth and CH_4 emissions from the swamp columns because of the small CH_4 emission values. The CH_4 emissions from taiga bog microcosms were approximately 0.02-19.16 mg C m^{-2} day^{-1} (Funk *et al.* 1994). The low CH_4 production in tropical peat soil likely occurred because most of the supply and the highest quality of decomposable organic matter are restricted to the peat surface and CH_4 oxidation by methanotrophic bacteria under oxic conditions may exceed gas production in a deeper anoxic peat profile (Brady 1997). Many tropical peatlands are covered by forest, in contrast to temperate peatlands which are commonly covered by sedges and moss (Andriesse 1988). Woody tropical peat contains higher levels of recalcitrant materials (e.g., lignin). Wood contain a higher lignin content with less decomposable C compared with cellulose. Williams and Yavitt (2010) reported that the biochemical compositions of lignin affect soil methanogenesis

Regression analysis of the CH_4 fluxes per hour and the water depths indicated a negative relationship (P < 0.05) (Figure 4.6). According to this equation, i.e., $y = 0.00003x^2 - 0.0024x + 0.0709$, the lowest CH_4 flux can be reached when the water depth is 40 cm below the soil surface. Methanogenesis may take place at a higher water depth causing higher emissions of CH_4 to the atmosphere. Roulet *et al*. (1992) and Nykanen *et al*. (1998) show that the critical water depth for high CH_4 emissions is approximately 10–20 cm. According to Wosten *et al.* (2006), CH_4 emissions are negligible at water depths more than 20 cm below the surface, and there is an increase in the CH_4 emission at water depths above 20 cm. There was no interaction effect between water depth and amelioration on the CH_4 fluxes, and there was no significant water depths and ameliorations effect on CH_4 emission, which was probably due to the high standard deviation within the treatments. Although there was no significant difference, a CH_4 emission reduction trend was observed as the water depth was lowered. The CH_4 emission measured at the 35 cm and 55 cm water depths were 32 and 12% lower, respectively, than those measured at the 15 cm water depth. According to Jungkunst *et al.* (2008), the CH_4 emissions from temperate forest in peat soil columns were reduced by approximately 3 and 8% at water depths of 20 cm and 40 cm, respectively, compared with 5 cm below the soil surface. The reduction in CH_4 emission due to the lowering of the water depth is probably related to the soil Eh. In our study, the Eh varied with the water depth and influenced the CH_4 emissions (r = 0.89). According to Minamikawa and Sakai (2005), the Eh decreased the total CH_4 emission from paddy field in mineral soil by 36%.

Generally, CH_4 emissions occur at values lower than -150 mV and increase with decreases in soil Eh (Wang *et al*. 1993). This study confirms that temperature controls CH_4 emission.

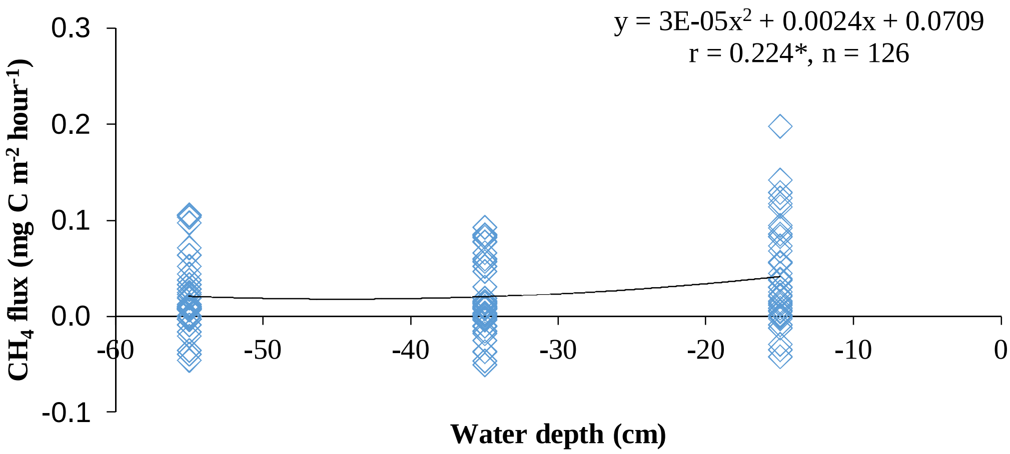


Figure 4.6. Relationship between water depths and CH_4 fluxes in peat soil (* indicates $P < 0.05$)

The annual CH_4 emissions from without ameliorant, the biochar+compost and steel slag+compost applications were approximately 3.5, 1.9 and 3.8 ton C ha^{-1} $year^{-1}$, respectively (Figure 4.7). The CH_4 emission from without ameliorant, the biochar+compost and steel slag+compost applications were approximately 40.0; 21.7 and 43.3 mg C m^{-2} $hour^{-1}$, respectively. The CH_4 emission from this study is higher compare with those found by ICCTF (2011) approximately 9-12 mg C m^{-2} $hour^{-1}$. The application of biochar+compost reduced the total CH_4 emissions by approximately 44%. By contrast, the CH_4 emission were slightly increased (6.7%) by the application of steel slag+compost. The CH_4 emitted from this treatment was approximately 0.2 ton C ha^{-1} $year^{-1}$ higher. The application of the biochar+compost reduced the CH_4 emissions in this study, similar to the biochar application results reported by Liu *et al.* (2011). The application of biochar+compost can reduce CH_4 emission probably because biochar provides better aeration, higher porosity, a larger surface area and makes the soil more favorable for methanotrophs compared with soils without biochar (Karhu *et al.* 2011). On the other hand, the steel slag+compost application stimulated CH_4 emissions, most likely because the electron acceptor activity associated with steel slag was not sufficient to accept all of the electrons released from the reduction process due to the high organic matter content (Lee *et al.* 2012). Compost and peat soil have high SOM contents. A similar result was shown by Susilawati *et al.* (2015), i.e., no significant reduction in CH_4 emissions, likely because the low rate of steel slag applications did not provide sufficient electron acceptors. A mechanism used to decrease CH_4 emissions is the addition of electron acceptors such as iron materials, which influence the sequential soil Eh reactions. The electron acceptors are ordered according to their Eh, and the substrate is used at lower concentrations by electron acceptors with a higher Eh (Lovley and Phillips 1988). In this study, the application of the steel slag+compost increased the soil Eh compared with biochar+compost treatment and without ameliorant. Furukawa and Inubushi (2002) reported that CH_4 production activity was decreased with high revolving furnace slag application (20-100 Mg ha^{-1}).

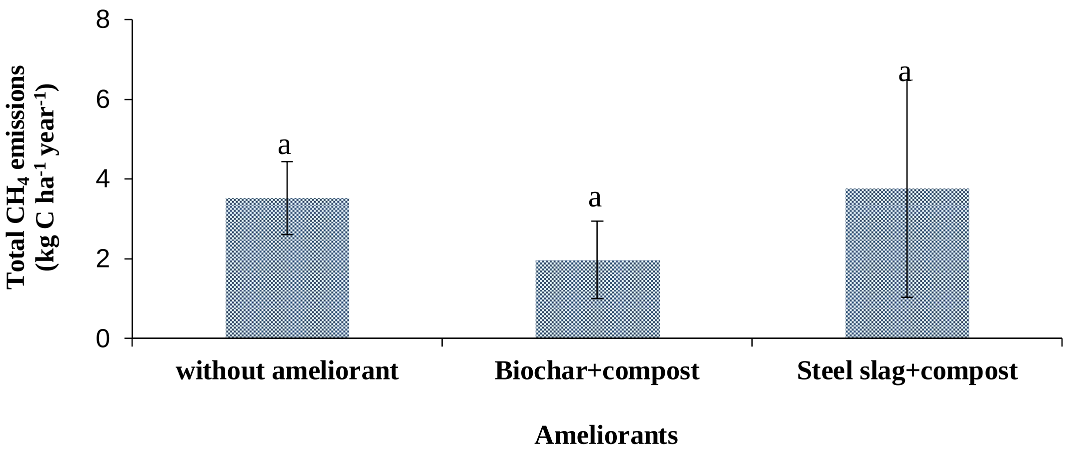


Figure 4.7. The CH_4 emission from different ameliorants at peat soil columns ($P < 0.05$)

3.3.3. Nitrous oxide emissions

The dynamic changes in the N_2O fluxes from peat soil columns are presented in Figure 4.8 for various water depths and ameliorants. The N_2O fluxes varied within and between treatments. At the first measurement at 15 cm water depths, the N_2O fluxes resulting from the application of steel slag+compost and biochar+compost were very high, likely because of the amelioration effect on soil pH and Eh. The first N_2O emission measurements were high (similar to CO_2 emissions), probably due to the high soil pH and Eh that occurred early in experiment. In this study, amelioration significantly ($P < 0.05$) affected the N_2O emission. After the first time measurement conducted in these treatment, N_2O fluxes at 35 and 55 cm water depths increased for next 5 measurements, i.e., until the end of the measurements. The fluxes from without ameliorant were mostly lower than those resulting from the application of ameliorants. At the first measurement, the N_2O fluxes at 15 cm water depths in the application of steel slag+compost and biochar+compost treatments were very high. In this study, the N_2O fluxes from all the treatments ranged from approximately 39-151 µg N m^{-2} $hour^{-1}$. According to Jungkunst *et al.* (2008), N_2O fluxes from temperate forest were approximately around 20-963 µg N m^{-2} $hour^{-1}$, based on soil columns measurements at different water depths. In addition, N_2O fluxes from tropical peat soil under secondary forest and paddy field on tropical peat soil in South Kalimantan were approximately 46 and 154 µg N m^{-2} $hour^{-1}$, respectively (Hadi *et al.* 2001).

Figure 4.9 shows that the N_2O fluxes a had nonlinear, (a quadratic) relationship with water depth; $y = -0.00001x^2 - 0.0002x + 0.0488$. Based on this equation, the maximum N_2O flux occurred when the water depth was 10 cm below the soil surface. This finding similar to that reported by Jungkunst *et al*. (2004) and Furukawa *et al.* (2005), i.e., that the peak N_2O flux was observed at water depth of 20 cm below soil surface. Jungkunst *et al*. (2008) also observed a quadratic function between water depths and N_2O fluxes. This indicates that the N_2O emitted during both nitrification and denitrification depends on the soil water content. The highest N_2O emissions are found at intermediate water depths, which allows for both aerobic and anaerobic conditions (Davidson *et al.* 2000). Denitrification rates in soil generally depend on the O_2 concentration, the NO_3^- concentration and the availability of easily metabolizable organic matter (Stepniewski and Stepniewska 1998). Lowering the water table generally increases the N_2O emission rate; however, this relationship is rather complex (Regina *et al.* 1996).

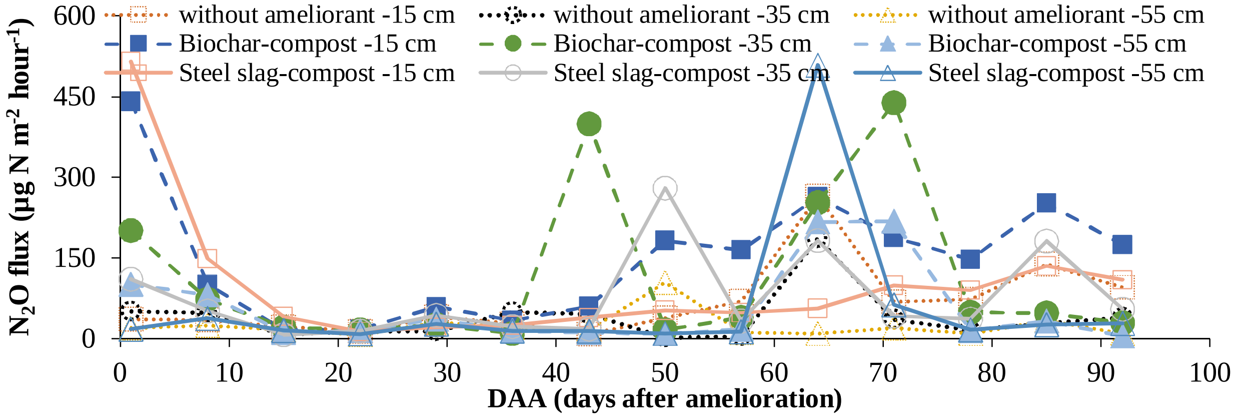


Figure 4.8. The changes in N_2O fluxes from 3 different water depths and ameliorations during the experimental period.

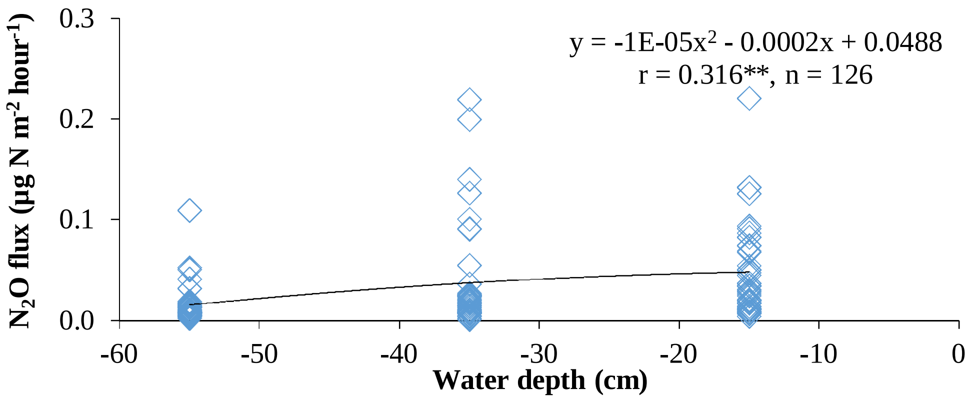


Figure 4.9. Relationship between water depths and N_2O fluxes in peat soil (** indicates $P < 0.01$)

The emissions of the N_2O from the biochar+compost and steel slag+compost treatments were higher than those from without ameliorant (Figure 4.10). The high N_2O emission from ameliorant treatments most likely occurred because of the increased availability of the nitrate substrates and the easily degradable organic matter for denitrification, which resulted from the compost application (Linn and Doran 1984; Dobbie *et al.* 1999). The availability of organic C is the main factor that influences denitrification under anaerobic conditions (Zou *et al.* 2005). Emissions of N_2O from the soil amended with biochar depend on the characteristics of the biochar, the addition of exogenous nitrogen and soil properties (Zhang *et al.* 2010). Singh *et al.* (2010) explained that the N_2O emission increased due to higher microbial activity and high labile N content of the biochar. In addition, soil pH increases in tropical acid soils supplied with composts derived from organic products results in (van der Watt *et al.,* 1991). The mechanism of increasing soil pH through the application of organic matter is not fully understood but likely occurs because of the specific adsorption of organic anions and the corresponding release of hydroxyl ions (Hue 1992). The higher N_2O emission from the steel slag+compost treatment is similar to the result with steel slag application result obtained by Huang *et al*. (2009); Liu *et al* (2012) and Singla and Inubushi (2013). The N_2O emissions were significantly enhanced by the addition of iron. Iron can affect the speciation and mobility of organic and inorganic substances in soils and subsequently alleviate the immobilization of fertilizer N accompanying the decomposition of incorporated crop residue with a high C/N ratio, making more mineral N available for nitrification and denitrification. In addition, the soil pH was increased by the steel slag application. According to Ali *et al.* (2009), the alkaline pH of steel slag contributed to the increase in soil pH. The biochar+compost and steel slag+compost treatments increased of the soil pH. This finding showed that the ameliorations increased the soil pH. The mechanisms of N_2O production resulting from soil amelioration are governed by nitrification, nitrifier denitrification and denitrification (Khalil *et al.* 2004). The highest annual N_2O emission was measured in the biochar+compost treatment followed by the steel slag+compost treatment and without ameliorant, i.e., approximately 9.4; 6.9 and 3.8 ton N ha^{-1} year $^{-1}$, respectively (Figure 4.10). The application of Biochar+compost and steel slag+compost to peat soil significantly stimulated N_2O emissions by approximately 5.6 and 3.0 ton N ha^{-1} year $^{-1}$, respectively. In this study, N_2O emission from without ameliorant, biochar+compost and steel slag+compost were approximately 44.38; 107.31 and 78.77 mg N m^{-2} $hour^{-1}$, respectively. The range of N_2O emission from this study was quite higher compare the result from Sakata *et al.* (2015), i.e., approximately 272 µg N m^{-2} $hour^{-1}$ (Table 1.1). The difference range of N_2O emission from these studies could be happened most likely because of many factors, e.g., rain, temperature, fertilization, irrigation, heavy metal accumulation, pH, organic matter content (Khalil *et al.* 2003), degree of peat maturity and experimental conditions.

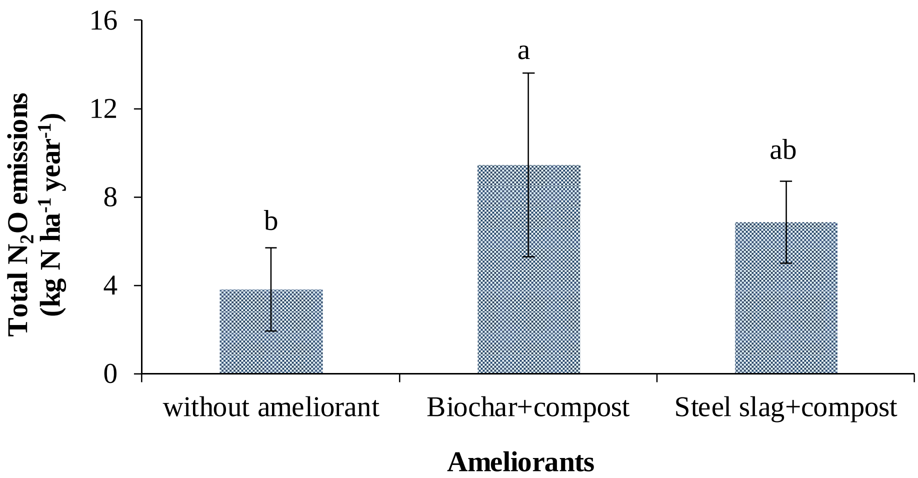


Figure 4.10. The N_2O emission from different ameliorants at peat soil columns (P < 0.05)

This study yielded different results from Susilawati *et al.* (2015). In that study, the N_2O emissions were significantly reduced by steel slag application after second seasons (Table 4.3).

There was a statistical decrease in N_2O emissions in the rice plantation from steel slag applications; the reduction was approximately 39–49%. Lower N_2O emissions were found after steel slag application at an 8 Mg ha^{-1} rate compared with the control according to Wang *et al.* (2015). The reduced N_2O emissions can be caused by an increase in the iron (III) oxide concentration, suppressing microbe activities, including N_2O production (Noubactep 2011). Steel slag is high in iron, which is an important oxidant for biological and chemical reactions that use oxidizing or reducing agents. Nitrite can be reduced by the presence of iron oxide at a near-neutral pH, and the end product is N_2O and NO as an intermediate (Van Cleemput and Baert 1983; Van Cleemput 1998). The reduction of nitrite will affect the global production of nitric oxide (NO) and N_2O. Kampschreur *et al.* (2011) described the chemical conversions as follows:

$NO_2^- + Fe^{2+} + 2H^+ \rightarrow Fe^{3+} + NO + H_2O$ (1)

$NO + Fe^{2+} + 1H^+ \rightarrow Fe^{3+} + 0.5N_2O + 0.5H_2O$ (2)

Abiotic denitrification with iron (II) can occur during nitrite accumulation. High nitrite concentrations increased nitrogen availability, especially when fertilizers are applied. During anoxic, an iron (II)/iron (III) reduction may induce biological and chemical nitrate and nitrite reduction to NO and N_2O (Kampschreur *et al.* 2011). Biological iron oxidation with nitrate releases small amounts of N_2O accumulation, and under strong anaerobic conditions, N_2O is further reduced to N_2 as an end product (Nielsen and Nielsen 1998; Granli and Bockman 1994). These reactions indicate that when there are higher iron applications to soil or higher iron contents in soil, this element will influence N_2O emissions. The response in terms of N_2O emissions reduction depends on the physicochemical properties of the soil. The availability of iron in soils is influenced by the type of parent material and the land use in different farming systems.

Table 4.3. Seasonal N_2O emissions at two different paddy field sites during rice growing seasons (Susilawati *et al.* 2015)

Sites	Application of steel slag	N_2O DS (g N ha^{-1} season^{-1})		N_2O RS (g N ha^{-1} season^{-1})	
Jakenan	Control	38.75	a	19.31	bc
	Steel slag 1 Mg ha^{-1}	30.57	a	13.84	c
	Steel slag 2 Mg ha^{-1}	-		12.78	c
Wedarijaksa	Control	45.99	a	46.03	a
	Steel slag 1 Mg ha^{-1}	41.43	a	33.70	ab
	Steel slag 2 Mg ha^{-1}	-		28.37	abc
	ANOVA				
	Sites	ns		***	
	Application of steel slag	ns		*	
	Sites*steel slag application	ns		ns	

4.3.4. Redox potential (Eh) and pH

At the first measurement, the Eh from in all treatments had a positive value (Figure 4.11). The different treatments had distinct effects on the Eh starting from the second measurement until the end of the experiment. Highly reducing conditions developed at the 15 cm water depth in the three amelioration treatments. In the 15 cm water depth treatment, the first Eh value was positive and then gradually decreased. During the entire experimental period, the Eh values measured at the 15 cm water depth ranged from approximately +356 mV to -138 mV. At the lower water depths (35 and 55 cm), the Eh increased to more or less constant values ranging from +202 to +479 mV which were significantly higher than those measures at 15 cm water depths. The clear differences in Eh observed in this study likely occurred because of the lowering of water. There was a relationships between the Eh and water depth (r = 0.89, n = 42, $P < 0.01$). On the other hand, under anaerobic conditions, oxygen in soil decreases and then the Eh gradually decreases as a result of the biochemical activity of numerous facultative and obligate anaerobes that use NO_3^-, Fe(III), Mn(IV) compounds and SO_4^{2-} as terminal electron acceptors. The production of CO_2 decreases linearly with Eh (Włodarczyk *et al.* 2002).

The Eh was correlated with the three gases, i.e., CO_2, CH_4, N_2O (Table 4.4). There was a positive correlation between the Eh and the CO_2 fluxes in all treatments. The Eh and CO_2 fluxes were positively and negatively correlated in without ameliorant and the biochar+compost treatment, respectively, at a water depth 55 cm. There were correlations between the Eh and CH_4, N_2O fluxes in all treatments. In addition, there was a correlation between the Eh and the N_2O fluxes in without ameliorant at water depth of 55 cm.

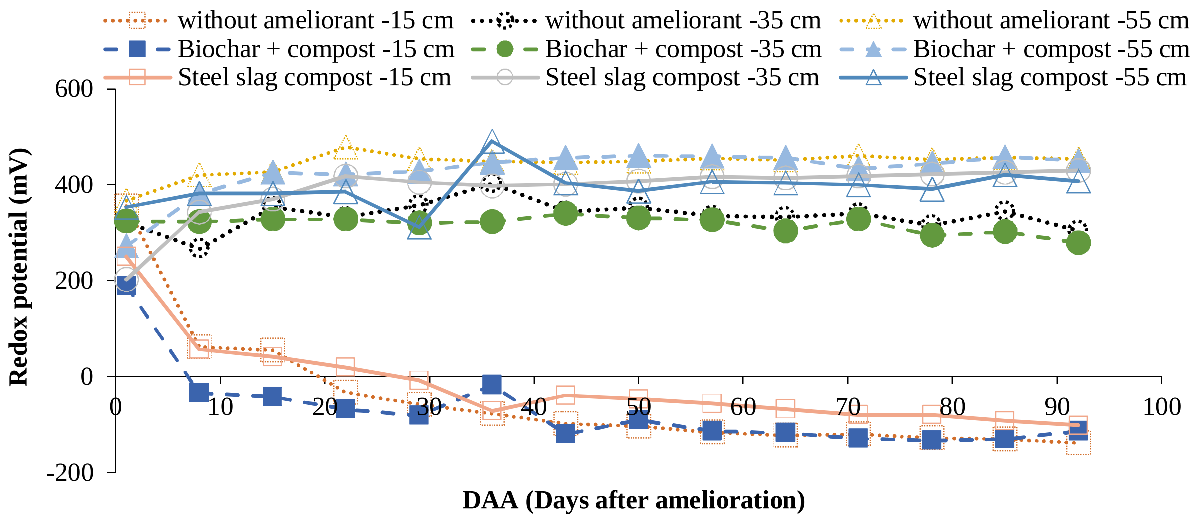


Figure 4.11. The changes in redox potential from 3 different water depths and ameliorations during the experimental period.

At the first measurement, the soil pH in ameliorant treatments was higher than that measured in the treatment of without ameliorant for all water depths (Figure 4.12). Thereafter, the soil pH of without amelioration was consistently lower than that measured in the biochar+compost and steel slag compost treatments at all water depths. The soil pH values measured after amelioration increased slightly throughout the measurement period. The soil pH in without ameliorant ranged from approximately 2.98 to 3.40 for the water depths and that measures in the biochar+compost and steel slag compost treatments was approximately 2.94 to 4.41 and 3.21 to 4.73, respectively. Soil pH affects the dynamics of carbon. The intensity of soil respiration is closely related to the decomposition of soil organic carbon and soil (Silva *et al.* 2008). Moreover, the compost in this study contained lime. Lime is used in agriculture to increase soil pH (West and McBride 2005). Steel slag contains high iron content, which changed the pH to alkaline; therefore the increase in soil pH may have enhanced the activity of soil microorganisms, and accelerated the decomposition of organic matter.

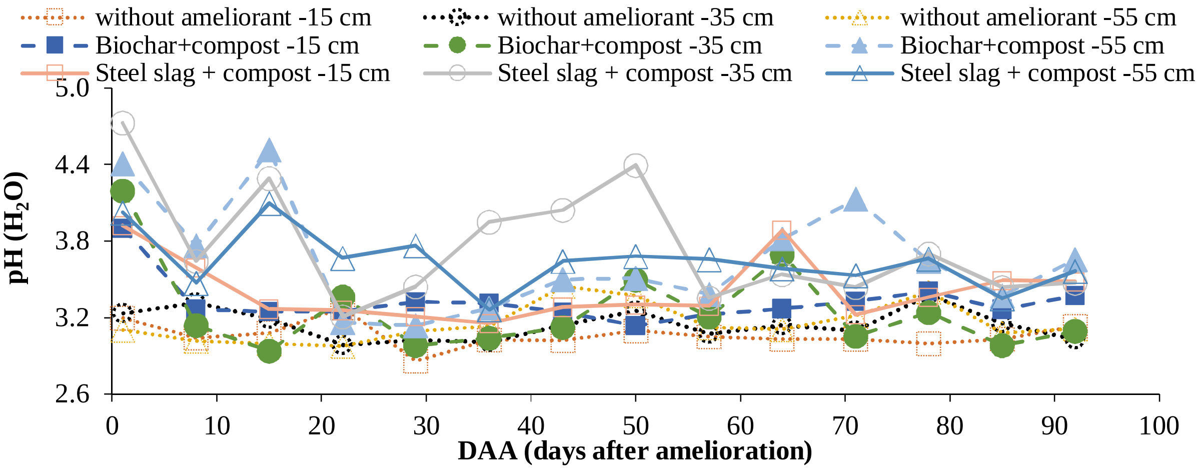


Figure 4.12. The changes in pH from 3 different water depths and ameliorations during the experimental period.

There were positive correlations between pH and the CO_2 and N_2O fluxes with pH (P < 0.01) (Table 4.4). Correlations were observed between pH and the CO_2 fluxes in the Biochar+compost treatment at 35 cm water depth and in without ameliorant at 55 cm water depth. Correlations were observed between pH and CH_4 fluxes in steel slag+compost treatment at 35 cm water depth and between pH and N_2O fluxes in without ameliorant at 35 and 55 cm water depths.

4.3.5. Soil and water temperature

There was a decreasing trend of soil and water temperature in all treatments over the experimental period (Figure 4.13a and 4.13b). The soil and water temperatures ranged from 25.7 - 32.2^{0}C and 27.1 - 33.1^{0}C, respectively. The average soil temperature was 29^{0}C. At depths of 15, 35 and 55 cm, the average soil temperature was 29.2, 29.0 and 28.9^{0}C and the average of water temperatures were 30.0, 29.8 and 29.2^{0}C, respectively. The soil and water temperature pattern were similar across all treatments. The water and soil temperature decreased at 30 DAA. There was no significant difference between treatments. The soil temperature was strongly correlated with the CO_2, CH_4 and N_2O fluxes from all treatments (Table 4.4). According to Furukawa *et al.* 2005, soil temperature controls the biological reaction in the soil and then influences gas production. This study showed that lower water depths resulted lower in soil and water temperatures. An increase in soil temperature could lead to increased CO_2 and CH_4 fluxes from peatlands (Williams and Crawford 1984). In this study, there was a strong correlation between soil temperature and CO_2 emission (P < 0.01) (Table 4.4). The soil and water temperatures were strongly correlated with CH_4 emission (Table 4.3). The correlation between temperature and CH_4 emission is more complicated than that for CO_2 emission due to the differing responses of CH_4 production and consumption processes in peat soil (Moore and Dalva 1993). Dunfield *et al.* (1993)

showed that CH_4 production and consumption reached an optimum at 25-30^{0}C in temperate peat. There was a correlation between the soil temperature and the CH_4 fluxes in the steel slag+compost treatment at a depth of 55 cm. The water temperature was only correlated with the CH_4 fluxes (Table 4.4).

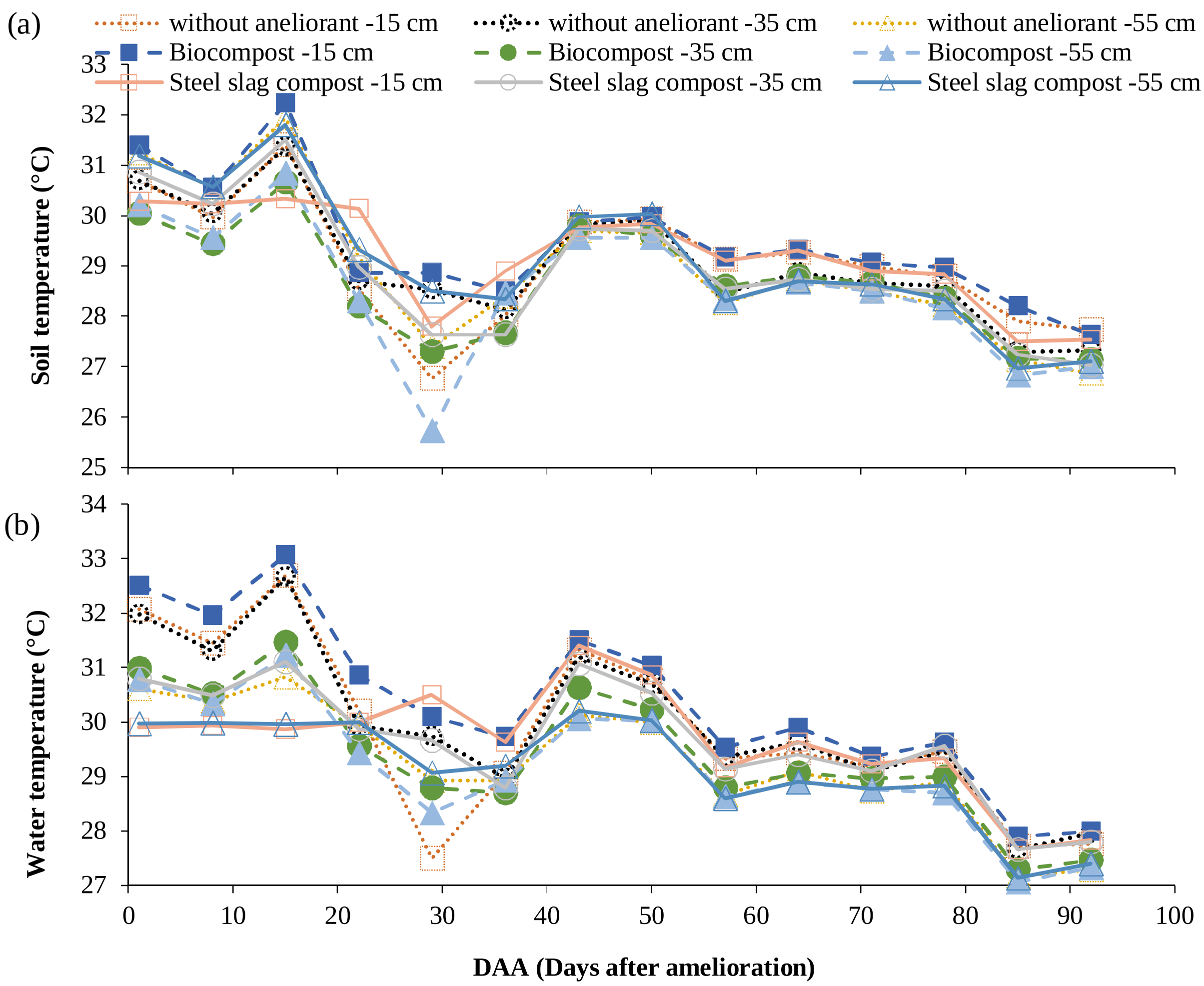


Figure 4.13. The changes in soil (a) and water (b) temperature from 3 different water depths and ameliorations during the experimental period.

Table 4.4. Correlation co-efficient (r) CO_2, CH_4 and N_2O fluxes with soil Eh, pH and temperature from each (n =14) and whole treatments (n = 126)

Parameters	15 cm			35 cm			55 cm			Whole treatments
	without ameliorant	Biochar+ compost	Steel slag+ compost	without ameliorant	Biochar+ compost	Steel slag+ compost	without ameliorant	Biochar+ compost	Steel slag+ compost	
					CO_2					
Eh (mV)	-0.285	0.280	-0.013	0.143	-0.042	-0.108	0.828**	-0.736**	-0.168	0.47**
pH	-0.290	-0.081	0.255	0.142	0.538*	0.487	0.688**	0.359	0.287	0.66**
Soil temperature (°C)	0.159	0.036	-0.023	0.166	0.195	0.190	0.438	0.343	0.266	0.34**
Water temperature (°C)	0.049	-0.009	-0.049	0.202	0.084	0.039	0.147	0.185	0.167	0.12
					CH_4					
Eh (mV)	0.004	0.113	0.026	-0.031	0.065	-0.405	0.475	0.307	-0.392	-0.24*
pH	0.261	-0.357	-0.648**	0.177	0.592*	0.026	0.003	-0.477	0.437	0.14
Soil temperature (°C)	-0.100	-0.106	0.220	0.032	0.449	-0.013	0.420	-0.358	0.736**	0.46**
Water temperature (°C)	0.070	-0.034	0.323	-0.069	0.346	-0.034	0.060	-0.344	0.513	0.32**
					N_2O					
Eh (mV)	-0.343	-0.175	-0.019	0.413	0.282	-0.201	0.760**	-0.020	0.073	-0.43**
pH	-0.167	0.002	0.350	0.667**	0.186	0.431	0.702**	0.263	-0.069	0.45**
Soil temperature (°C)	-0.195	-0.039	0.093	0.083	0.314	0.099	0.212	-0.023	-0.086	0.28**
Water temperature (°C)	-0.412	-0.034	0.133	-0.115	0.196	0.018	-0.128	-0.089	-0.073	0.07

* and ** denote significant at $p < 0.05$ and 0.01, respectively

4.3.6. Soil moisture

Soil samples to measure soil moisture were taken from 2 different depth, i.e., upper layer with the depth 0-10 cm from soil surface and lower layer with the depth 40-50 cm from soil surface. In all the treatments showed lower soil moisture in upper layer than in lower layer (Figure 4.14.). In upper layer, soil moisture were approximately around 70-74%, while in lower layer, soil moisture ranged approximately 75-87%. The lowest soil moisture in both of soil depth was found in the -55 cm water depth below soil surface, followed by -35 cm and -15 cm water depth below soil surface. Soil moisture effect the diffusion of soluble substrates at lower soil water content whilst at higher soil moistures diffusion of oxygen can become constrained; both of which are limiting to soil microbial respiration (Skopp *et al.* 1990).

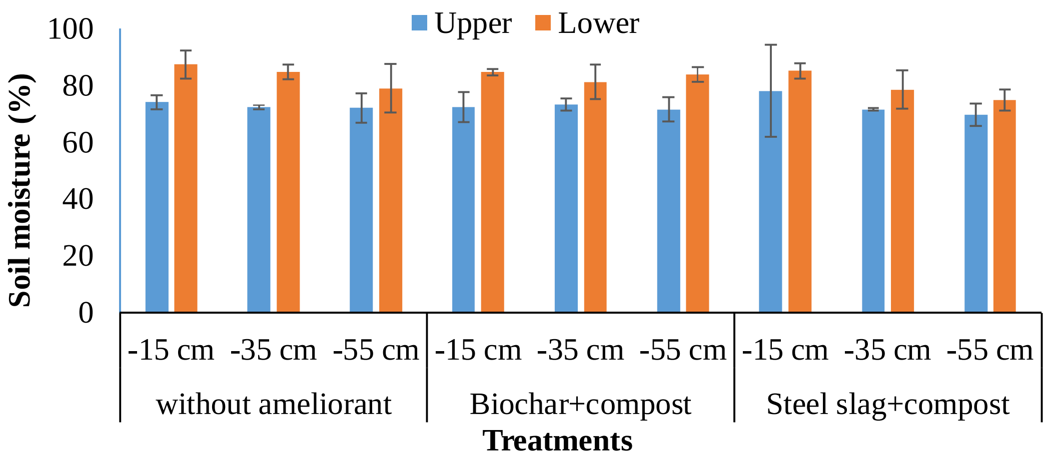


Figure 4.14. The soil moisture from 2 different soil depth under 3 different water depths and ameliorations during the experimental period.

There were linear correlation between soil moisture and water depth in different of soil layer (Figure 4.15.). Deeper water depth resulted lower soil moisture. Drainage create aerobic conditions and increase in the redox potentials are the condition that favour microbial activity and nitrogen mineralization (Ueda *et al.* 2000, Jali 2004). Lowering water depth increase peat soil aeration, then optimising microbial oxidation of organic matter to release of CO_2 to the atmosphere and reduce CH_4 emission. CH_4 is a characteristic product of organic matter breakdown under anaerobic peat, and gas production is highest when the water depth is near or at the peat surface and less decomposed litter becomes available for anaerobic decomposers (Conrad 1989).

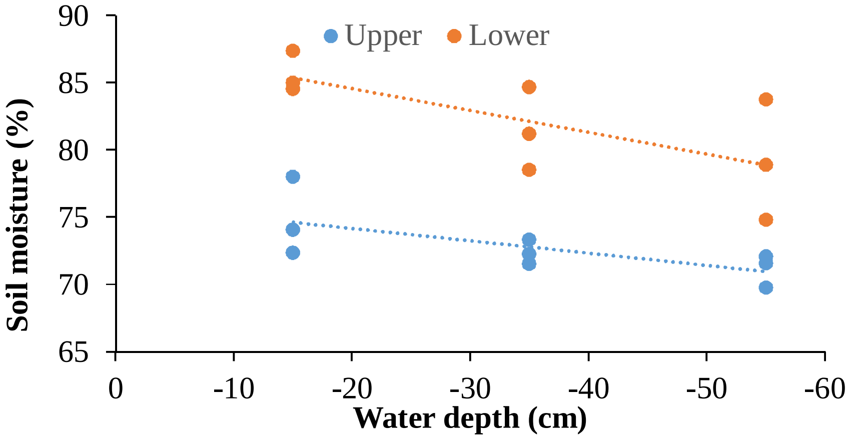


Figure 4.15. Correlation between soil moisture from 2 different soil depth and water depth

4.3.7. Total GHG emissions

There was no interaction effect between water depth and amelioration on the CO_2 emissions (Table 4.5). The effect of each treatment on CO_2 emissions was independent. The CO_2 emissions were significantly ($P < 0.05$) and highly significantly ($P < 0.01$) affected by water depth and amelioration, respectively. The annual emissions of CO_2 eq from all treatments ranged from 10.7-35.8 ton CO_2 eq ha^{-1} $year^{-1}$. There was no interaction effect between water depth and amelioration on the CH_4 emissions. Moreover, there was no significant water depth and amelioration effect on the CH_4 emissions. The annual CH_4 emissions from all treatments ranged from 0.04-0.38 ton CO_2 eq ha^{-1} year^{-1}, respectively. There was no interaction effect between water depth and amelioration on N_2O emissions. There was a significant ameliorations effect on the N_2O fluxes, but no significant water depths effect. The annual N_2O emissions from all treatments ranged from 1.01-6.15 ton CO_2 eq ha^{-1} year^{-1}, respectively. Amelioration significantly increased the total N_2O emissions.

There was no interaction effect between water depth and amelioration on the total GHG emissions. In addition, there was no significant water depths effect on the total GHG emissions from the peat soil columns, but there was a significant amelioration effect. The total GHG emissions from the different water depths and soil ameliorants added to peat soil were ranged from approximately 13.6-40.2 ton CO_2 eq ha^{-1} year^{-1}. The total GHG emission from without ameliorant was approximately around 18.1 ton CO_2 eq ha^{-1} year^{-1} and those from Biochar+compost and steel slag+compost treatments were approximately around 36.4 and 24.8 ton CO_2 eq ha^{-1} year^{-1}, respectively. The total GHG emissions from biochar+compost treatment were almost double than those from without ameliorant. The biochar+compost and steel slag+compost application significantly stimulated total GHG emissions from peat soil by approximately 18.3 and 6.7 ton CO_2 eq ha^{-1} year^{-1}, respectively. The contribution of CO_2, CH_4 and N_2O to the total GHG emissions from the peat soil column were approximately 87; 0.5 and 12.6%, respectively. Total GHG emissions expressed as CO_2 eq facilitate access to the most acceptable technologies for reducing GHG emissions without having to separate each gas. In this study, ameliorations added to peat soil stimulated the total GHG emissions compared the treatment without ameliorants. Therefore, adding ameliorants to peat soil to enhance soil fertility should be more considered due to their effect on GHG emissions.

Table 4.5. CO_2, CH_4, N_2O and total GHG emissions from different water depths and ameliorants at peat soil columns

Treatments		CO_2 emission				CH_4 emission				N_2O emission				Total GHG emissions	
Ameliorants	**Water depth (cm)**	**mg C m^{-2} hour^{-1}**		**ton CO_2eq ha^{-1} year^{-1}**		**mg C m^{-2} hour^{-1}**		**ton CO_2eq ha^{-1} year^{-1}**		**µg N m^{-2} hour^{-1}**		**ton CO_2eq ha^{-1} year^{-1}**		**ton CO_2eq ha^{-1} year^{-1}**	
without ameliorant	15	33	b	10.72	b	0.049	a	0.29	a	67	a	2.74	a	13.6	c
	35	53	ab	17.06	ab	0.043	a	0.26	a	39	a	1.60	a	18.8	bc
	55	64	ab	20.43	ab	0.029	a	0.17	a	25	a	1.01	a	21.5	bc
Biochar + compost	15	76	ab	24.33	ab	0.012	a	0.07	a	151	a	6.15	a	30.5	ab
	35	110	a	35.33	a	0.034	a	0.20	a	116	a	4.74	a	40.2	a
	55	112	a	35.87	a	0.022	a	0.13	a	57	a	2.32	a	38.3	a
Steel slag + compost	15	45	ab	14.52	ab	0.063	a	0.38	a	100	a	4.08	a	18.8	bc
	35	78	ab	25.08	ab	0.007	a	0.04	a	77	a	3.16	a	28.3	abc
	55	76	ab	24.31	ab	0.059	a	0.35	a	58	a	2.35	a	26.8	abc
	ANOVA														
	Amelioration	**		**		ns		ns		*		*		**	
	Water depth	*		*		ns		ns		ns		ns		ns	
	Interaction	ns		ns		ns		ns		ns		ns		ns	

P < 0.05, ***P* < 0.01, ns = not significant. Values in each column are means of three replicates Different letters in the same column indicate significant differences between means at P = 0.05 according to Tukey's HSD test

4.3.8. Changes in carbon stock and net carbon budget associated with water depths and ameliorations

The carbon budget of agroecosystems is important in the global terrestrial C cycle (Pan *et al.* 2004). Increasing agricultural soil C stocks has been suggested as an important potential measure to sequester CO_2 from the atmosphere (Paustian *et al.* 1998). Soils are the second largest terrestrial carbon (C) reservoir (Jacinthe *et al.* 2002). The net carbon budget is typically estimated from soil organic carbon (SOC) measurements (Mosier *et al.* 2005; Robertson *et al.* 2000; Shang *et al.* 2010).

There was no difference among the treatments on C stock before and after the treatments. The highest C stock after the treatments was found in biochar+compost application, followed by steel slag+compost and without ameliorant and application approximately 1.26, 1.23 and 1.19 kg C column^{-1}, respectively. Based on the water depth, the highest C stock after the treatments was found in -35 cm below soil surface and followed by -15 cm and -55 cm below soil surface approximately 1.39, 1.30 and 1.00 kg C column^{-1}. C stock after the treatments was lower compare than before the treatments likely occur because there was peat decomposition resulted CO_2, CH_4 and N_2O emission (Table 4.5). Soil carbon stocks (per unit area) are estimated as the product of carbon concentration (% C), bulk density (g cm^{-3}), and soil volume (m^{-3}) (Warren *et al.* 2012). These properties cannot be measured directly from satellite or airborne sensors, and therefore rely on intensive field sampling for data acquisition. Bulk density and C concentration can vary spatially and throughout the vertical peat profile (Page *et al.* 2004). Therefore, in the future study multiple measurements of carbon concentration and bulk density from samples taken at various depths in the soil profile are needed to accurately determine the soil carbon stocks (Donato *et al.* 2011; Kauffman *et al.* 2011; Murdiyarso *et al.* 2010).

In all the treatments, C stock was higher than total GHG emission (Table 4.6). The net CO_2-C exchange between the atmosphere and terrestrial were approximately 0.76-1.23 kg CO_2-C column^{-1} 92 days^{-1}. There was no significant difference on net carbon among the treatments. Net CO_2 exchange between the atmosphere and terrestrial systems represents the balance between C inputs by autotrophic fixation and outputs by heterotrophic oxidation of organic material. In this study, net carbon was determined from changes in topsoil organic C and total CO_2-C fluxes.

Table 4.6. C stock, total GHG emission and net carbon under different water depths and ameliorations

Treatments		**C stock kg (C column-1)**			**Total GHG emission**	**Net Carbon**
		Before treatments	**After treatments**	**Δ c stock**	**(kg CO_2-C column^{-1} 92 day^{-1})**	**(kg CO_2-C column^{-1} 92 day^{-1})**
without ameliorant	15 cm	1.75	1.26	-0.49	0.10	1.15
	35 cm	1.80	1.37	-0.43	0.14	1.23
	55 cm	1.76	0.96	-0.80	0.15	0.81
Biochar+ compost	15 cm	1.87	1.35	-0.52	0.23	1.12
	35 cm	1.85	1.41	-0.45	0.29	1.11
	55 cm	1.88	1.03	-0.85	0.27	0.76
Steel slag+ compost	15 cm	1.80	1.29	-0.51	0.14	1.15
	35 cm	1.84	1.41	-0.44	0.21	1.20
	55 cm	1.82	1.00	-0.82	0.19	0.81

4.4. Conclusions

This study quantified the effects of water depth and soil amelioration on greenhouse gas emissions from peat columns. Less CO_2 was emitted lower when the water depth was near the soil surface. Conversely, a deeper water depth resulted in a slight decrease in the CH_4 emissions. However, the highest N_2O emissions were measured at intermediate water depths. The soil pH, Eh and temperature were associated with the three gases (CO_2, CH_4 and N_2O) emitted from the tropical peat soil. The biochar+compost and steel slag+compost treatments increased the CO_2 and N_2O emissions from the peat soil columns. Long-term experiments should be developed to monitor changes that occur over time in response to amelioration at various water depths. It might be better to apply ameliorants at higher rates to reach a sustainable reduction in GHG emissions but it should consider the applicability to be used by the farmer. These experiments should be conducted in peat soil and peat soil-crop systems in peat soil to determine how GHG emissions from these types of treatments can be reduced.

CHAPTER 5

GENERAL DISCUSSION AND CONCLUTIONS

5.1 General discussion

Addressing the global challenges of climate change, food security, and the increase of human population requires enhancing the adaptive capacity and mitigation potential of agricultural sector. The agricultural sector is one of the sectors that currently responsible for global GHG emissions and is also a key driver of deforestation due to food demand. Global warming continues to dominate the world's science and policy agenda on global change. One fundamental concern is the impact of this climate change on water supply. Rice production requires large amounts of water to produce more rice. On the other hand, in the future was expected that rice production will face water competition and water scarcity. To reduce water use in irrigated lowland rice, water-saving regimes can be introduced, such as alternate wetting and drying (AWD), intermittent irrigation, mid-season drainage. Reducing water from rice field can save the water as well as reduce environmental burden such as CH_4 emission, but it sometimes has side effects such as increasing N_2O emissions. The trade-off relationship of N_2O and CH_4 production in rice soils makes it a real challenge to reduce the production of one gas but not to increase the production of the other. A better understanding of this relationship is needed in order to be able to possibly mitigate the emission of these important greenhouse gases through changes in agricultural practices. The approaches to save the water and minimize the environmental burden should allow rice production to be maintained or increased because there is high demand on food crop as staple food.

There are important factor in soil that control CH_4 and N_2O production from rice fields. The content of soil oxidants used as electron acceptors for organic matter degradation contributes significantly to CH_4 and N_2O production. The reduction of various oxidants in homogeneous soil suspensions occurs sequentially at corresponding soil redox potential values (Ponnamporuma 1972). Fluctuation of soil water content determines soil aerobic and anaerobic conditions, which can be characterized by Eh. In flooded rice soil, there are two aerobic/anaerobic interfaces—flooded soil surface layer maintained by O_2 dissolved in the standing water, and plant rhizosphere maintained by O_2 diffusing through the rice plant. These aerobic/anaerobic interfaces control many redox reactions including nitrification, denitrification, cycling of iron and manganese compounds, sulphate reduction and sulphide oxidation, and CH_4 formation and oxidation.

Simultaneous mitigation options are different for CH_4 and N_2O emission since the production of these two gases take place under contrasting conditions, thus minimizing one gas should not increasing the emission of the other. The trade-off both of the emission should be well prepared for a balanced set of mitigation options, which optimize the emission trade-off in minimum cumulative radiative forcing of the two gases on GWP, thus having a lowest possible greenhouse effect. Reducing water from field not only could mitigate GHG emission but also could save the water. This study showed that the practice of AWD and site specific of AWD (S-AWD) produce no significant yield difference to CF and save the water. Water-saving techniques can reduce GHG emissions in a given area of rice land, but in most cases, the water saved will then be used to irrigate more rice land or new crops in future seasons. Subsequently, emission savings are offset by emissions created on newly irrigated land. It means that rice production will increase. In the community and as a social concern, the relationship among water users improved, especially within an irrigation unit, because water had become available not only upstream but also downstream.

Growing rice in continuously flooded fields has been taken for granted for centuries, but water crisis may change the way rice is produced in the future. Technologies that save water for rice and increase productivity of a post-rice crop will be more acceptable to farmers. Assuming that AWD or S-AWD have been successfully introduced by a significant number of farmers to guide the irrigation of their rice crop, there is still the issue of providing mechanisms for the continued spread of AWD or S-AWD to be accepted on a larger scale in Indonesia. One difficulty in communicating about water savings itself is that this term carries different meanings to different people. The meaning is often dependent on the scales of interest: frequency, timing and volume of application, field preparation to control percolation and seepage and to capture rain, fertilizer use, pest control and more. Spread out the technology to the farmer is need more effort since implementing AWD or S-AWD require a coordinated approach for water scheduling in irrigation areas. Irrigation in each regions is managed in many different ways. In some cases, farmers are involved in decision-making processes on water scheduling, farmers and the pump owner discuss the irrigation schedule and payment arrangement before the onset of the irrigation season. In other cases, farmers who have their own water pump, are not formally organized and often independently implements the entire irrigation schedule by themselves. Thus, farmers with a bigger holding seem to be able to run the risk of losing some yield by trying AWD, who own a pump, greater economic stability and are able to control irrigation as prescribed by AWD.

Bouman and Tuong (2001) summarized 31 field experiments on AWD and they found the yield reductions of 0–70% in AWD treatments compared with continuously flooded controls in 92% of the experiments. To maintain or even increase the rice production. Another approaches could be done such as soil amelioration. Soil amelioration is the way to improve the soil quality to support the live of the plant and to enhance the production. Steel slag could be used as an oxidizing agent to suppress CH_4 emissions from rice fields. Electron acceptors such as NO_3^-, Mn^{4+}, Fe^{3+} and SO_4^{2-} can decrease CH_4 production because of inhibitory and competitive effects with different microorganisms for common electron donors (Jakobsen *et al*. 1981; Achtnich *et al*. 1995). Biochar, charcoal from biomass that has been pyrolysed in a zero or low oxygen environment, owing to its inherent properties, scientific consensus exists that application to soil at a specific site is expected to sustainably sequester carbon and concurrently improve soil functions. A mechanism of using soil amelioration used to decrease CH_4 emissions is the addition of electron acceptors, which influence the sequential soil Eh reactions. The electron acceptors are ordered according to their Eh, and the substrate is used at lower concentrations by electron acceptors with a higher Eh (Lovley and Phillips 1988). In this study, application of soil amelioration in mineral soil, i.e., steel slag application could slightly reduce CH_4 and N_2O emission in mineral soil although no significance difference. However, steel slag application as well as biochar and compost application in peat soil have tendency to stimulate GHG emission. Application of ameliorant in peat soil enhance GHG emission most likely because the electron acceptor activity associated with ameliorants were not sufficient to accept all of the electrons released from the reduction process due to the high organic matter content (Lee *et al*. 2012). Thus, it might be better to apply ameliorants at higher rates to reach a sustainable reduction in GHG emissions but it should consider the applicability to be used by the farmer. There is an urgent need for further experimental research with regard to long-term effects of soil amelioration on soil functions, as well as on the behaviour and fate in different soil types and under different management practices.

Recently, the peat-land utilization in Indonesia for agriculture has received much attention although the peat mostly contains of poor to very poor in nutrients for plant growth and organic materials (Sabiham 1988). Initially, all these peat deposits were covered with pristine peat swamp forest but, as a result of economic development, the peatlands have been subjected to intensive logging, drainage and conversion to plantation estates especially in Sumatra and Kalimantan (Rieley *et al.* 1996; Rieley and Page 2005). Mega Rice Project (MRP) initiated in 1995, disrupted the peatland ecosystem over an area of more than 1 million ha. MRP was constructed canals up to 30 m wide and length of approximately 4500 km. After the drainage, peatlands become susceptible to fire. Fires are most severe during El Niño periods, as in 1997/1998 when about 2.4–6.8 million ha of peatlands burnt in Indonesia (Page *et al.* 2002) after that El Niño reoccurred and burnt peatlands in Indonesia in 2002, 2006 and 2015.Furthermore, peatlands burnt once are more likely to burn again (Siegert *et al.* 2001; Cochrane 2003; Langner *et al.* 2007).

Expansion of agricultural land is widely recognized as one of the most significant human alterations to the global environment. On the other hand, there is increased competition for land, water, energy, and other inputs into food production. Consequently, the other way to meet increasing agricultural demands is looking toward the areas of arable land. The use of peat forest for agricultural activities has led to widespread declines in organic carbon (C) and hence in peat quality. The declines occur because, in such activities, the loss of organic-C is not offset by the gains of C through the deposition of biomass. The lowering of groundwater level by constructing drainage ditches in peatland areas was needed to convert the land to agriculture, but its impact on increasing greenhouse gas fluxes and soil decomposition resulted soil compaction was feared. Lowering of groundwater level by the drainage ditches in the peat lands contributed to greatly increased CO_2 fluxes, although CH_4 fluxes slightly decreased. However, the highest N_2O emissions were measured at intermediate water depths. In this study, soil amelioration in peat soil stimulate GHG emissions from the peat soil. Long-term experiments should be developed to monitor changes that occur over time in response to amelioration at various water depths. It might be better to apply ameliorants at higher rates to reach a sustainable reduction in GHG emissions but it should consider the applicability to be used by the farmer. These experiments should be conducted in peat soil and peat soil-crop systems in peat soil to determine how GHG emissions from these types of treatments can be reduced. Tropical forest peat land is the best land-use management in the peat lands to suppress carbon loss and greenhouse gas emission. Further long-term investigation is essential to do when lowering of groundwater level is needed to convert the land to agriculture to fulfil food demand.

5.2 General conclusion

From the above study, it could be concluded that:

1. This study results clearly delineated that appropriate management of irrigation or soil amelioration is likely a feasible approach if the approaches could eliminate the trade-off between CH_4 and N_2O production in rice soils and this condition makes more challenge on reducing the production of one gas but not to increase the production of the other.
2. Approaches that may contribute to reduce GHG emission from agricultural sector through the management of cultivated in mineral soil and peat soil should increase the yield then, it could be adopted, have a significant advantage for farmers. Further studies to verify the mitigation options should focus on feasibility for local farmers.

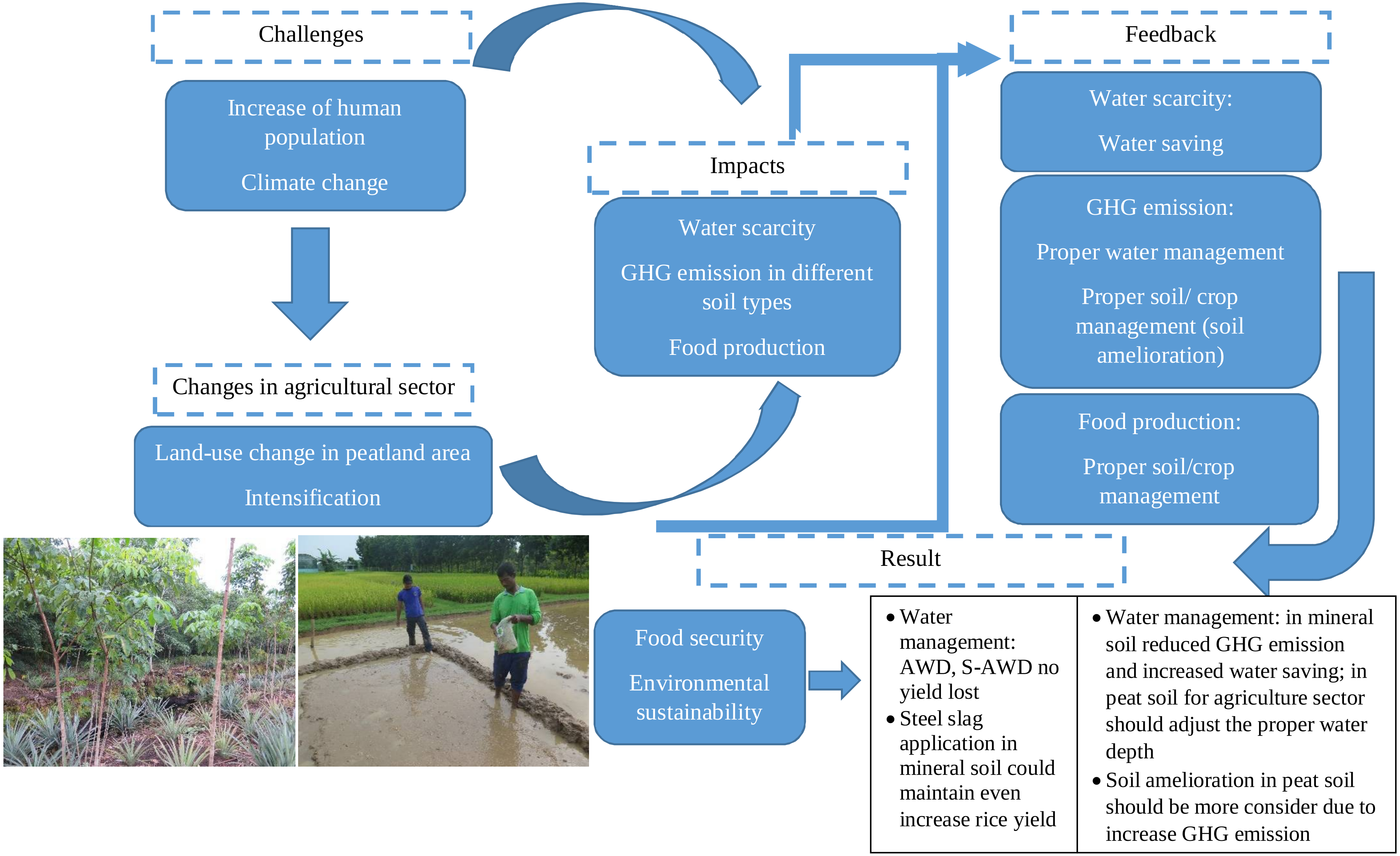


Figure 5.1. The overall conclusion from application of water managements and soil ameliorations in mineral soil and peat soil

REFERENCES

Abdullah K 2004: Rice production and paddy irrigation in the Asian monsoon region. Symposium on the International Network on Water Ecosystems and Paddy Fields (INWEPF)

Achtnich C, Bak F, Conrad R 1995: Competition for electron donors among nitrate reducers, ferric iron reducers, sulfate reducers and methanogens in anoxic paddy soil. *Biol. Fert. Soils*, **19**, 65–72.

Adhya TK, Patnaik P, Satpathy SN, Kumaraswamy S, Sethunathan N 1998: Influence of phosphorous application on methane emission and production in flooded paddy soils. *Soil Biol Biochem*., **30**, 177–181.

Adji FF, Hamada Y, Darung U, Limin SH, Hatano R 2014: Effect of plant-mediated oxygen supply and drainage on greenhouse gas emission from a tropical peatland in Central Kalimantan. *Soil Sci. Plant Nutr*., **60**, 216–230.

Aerts R, Ludwig F 1997: Water-table changes and nutritional status affect trace gas emissions from laboratory columns of peatland soils. *Soil Biol. Biochem*., **29**, 1691–1698.

Alberto MCR, Wassmann R, Hirano T, Miyata A, Kumar A, Padre A, Amante M 2009: CO_2/heat fluxes in rice fields: Comparative assessment of flooded and non-flooded fields in the Philippines. *Agricultural Forest Meteorol.*, **149**, 1737–1750.

Ali MA, Lee CH, Kim PJ 2008: Effect of silicate fertilizer on reducing methane emission during rice cultivation. *Biol. Fert. Soils*., **44**, 597–604.

Ali MA, Lee CH, Lee YB, Kim PJ 2009: Silicate fertilization in no-tillage rice farming for mitigation of methane emission and increasing rice productivity. *Agric. Ecosyts. Environ*., **132**, 16–22

Ali MA, Sattar MA, Islam MN, Inubushi K 2014: Integrated effects of organic, inorganic and biological amendments on methane emission, soil quality and rice productivity in irrigated paddy ecosystem of Bangladesh: field study of two consecutive rice growing seasons. *Plant Soil*, **378**, 239–252

Andriesse JP 1988: Nature and Management of Tropical Peat Soils. Soil Resources, Management and Conservation Service. FAO Land and Water Development Division. FAO Soils Bulletin 59.

Arah JRM, Smith K 1989: A steady-state denitrification in aggregate soils: a mathematical model. *J. Soil Sci*., **40**, 139–149.

Arai H, Hadi A, Darung U, Limin SH, Hatano R, Inubushi K 2014: A methanotropic community in a tropical peatland is unaffected by drainage and forest fires in a tropical peatlands. *Soil Sci. Plant Nutr*., **60**, 577–585.

Arif MAS, Houwen F, Verstraete W 1996: Agricultural factors affecting methane oxidation in arable soil. *Biol. Fertil. Soils*, **21**, 95–102.

Aulakh MS, Wassmann R, Bueno C, Kreuzwieser J, Rennenberg H 2001: Characterization of root exudates at different growth stages of ten rice (*Oryza sativa* L.) cultivars. *Plant Biology*, **3**, 139–148.

Baggs EM, Rees RM, Smith KA, Vinten AJA 2000: Nitrous oxide emission from soils after incorporating crop residues. *Soil Use Manage.*, **16**, 82-87.

Barker R, Dawe D, Tuong TP, Bhuiyan SI, Guerra LC 1999: The outlook for water resources in the year 2020: challenges for research on water management in rice production. *In*: Assessment and Orientation towards the 21st Century, Proceedings of 19th Session of the International Rice Commission, Cairo, Egypt, 7–9 September 1998. pp. 96–109. FAO, Rome,

Beare MH, Gregorich EG, St-Georges P 2009: Compaction effects on CO_2 and N_2O production during drying and rewetting of soil. *Soil Biol. Biochem.*, **41**, 611–621.

Bender M Conrad R 1995: Effect of CH_4 concentrations and soil conditions on the induction of CH_4 oxidation activity. *Soil Biol. Biochem.*, **27**, 1517–1527.

Berger S, Jang I, Seo J, Kang H, Gebauer G 2013: A record of N_2O and CH_4 emissions and underlying soil processes of Korean rice paddies as affected by different water management practices. *Biogeochem.*, **115**, 317–332.

Berglund Ö, Berglund K 2011: Influence of water table level and soil properties on emissions of greenhouse gases from cultivated peat soil. *Soil Biol. Biochem.*, **43**, 923–931.

Berglund Ö, Berglund K, Klemedtsson L 2010: A lysimeter study on the effect of temperature on CO_2 emission from cultivated peat soils. *Geoderma*, **154**, 211–218.

Bhuiyan SI 1992: Water management in relation to crop production: Case study on rice. *Outlook on Agriculture*, **21**, 293-299.

Bhuiyan SI, Sattar MA, Khan MAK 1995: Improving water use efficiency in rice irrigation through wet seeding. *Irrigation Science*, **16**, 1-8.

Bhuiyan SI, Tuong TP 1995: Water use in rice production: Issues, research opportunities and policy implications. Paper presented at the Inter-Center Water Management Workshop, 29-30 September 1995.

Bonkowski M 2004: Protozoa and plant growth: the microbial loop in soil revisited. *New Phytol.*, **162**, 616–631.

Bouman BAM, Tuong TP. 2001. Field water management to save water and increase its productivity in irrigated rice. *Agric. Water Manage*., **49**, 11-30.

Bouwman, AF 1991: Agronomic aspects of wetland rice cultivation and associated methane emissions. *Biogeochem.*, **15**, 65-88.

Brady MA 1997: Organic matter dynamics of coastal peat deposits in Sumatra, Indonesia. Ph.D. thesis, University of British Columbia, Vancouver, Canada.

Bremner JM 1997: Sources of nitrous oxide in soils. *Nutr. Cycling Agroecosyst.*, **49**, 7–16.

Bronson KF, Neue HU, Singh U, Abao EB 1997: Automated chamber measurements of methane and nitrous oxide flux in a flooded rice soil, I: Residue, nitrogen, and water management. *Soil Sci. Am. J.*, **61**, 981–987.

Cai ZC, Laughlin RJ, Stevens RJ 2001: Nitrous oxide and dinitrogen emissions from soil under different water regimes and straw amendment. *Chemosphere*, **42**, 113–121.

Cai ZC, Tsuruta H, Gao, M, Xu H, Wei C 2003. Options for mitigating methane emissions from a permanently flooded rice field. *Glob. Change Biol.*, **9**, 37–45.

Cai ZC, Xing GX, Yan XY, Xu H, Tsuruta H, Yagi K, Minami K 1997: Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. *Plant Soil*, **196**, 7–14.

Chapagain T, Yamaji E 2010: The effects of irrigation method, age of seedling and spacing on crop performance, productivity and water-wise rice production in Japan. *Paddy Water Environ*., **8**, 81–90.

Cheng W, Nakajima Y, Sudo S, Akiyama H, Tsuruta H 2002: N_2O and NO emissions from Chinese cabbage field as influenced by band application of urea or controlled-release urea fertilizers. *Nutr. Cycling Agroecosyst.*, **63**, 231–238.

Cicerone RJ, Oremland RS 1988: Biogeochemical aspects of atmospheric methane. *Global Biogeochem. Cycle*, **2**, 299–327.

Clymo RS 1983: Peat. *In*: Ecosystems of the World, 4B. Mires: swamp, bog, fen and moor, Regional Studies. Elsevier, Amsterdam, the Netherlands, pp. 159–224.

Clymo RS 1987: The ecology of peat. *Sci. Prog. Oxf.*, **71**, 593-614.

Coats B 2003: Global Rice Production. *In:* Rice, Origin, History, Technology and Production. Eds CW Smith and RH Dilday, pp. 247–270, John Wiley and Sons, Hoboken.

Cochrane MA 2003: Fire science for rainforests. *Nature*, **421**, 913–919.

Conrad R 1989: Control of methane production in terrestrial ecosystems. *In* Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. Dahlem Konferenzen (Andreae, M.O. and Schimel, D.S., Eds.), pp. 39-58. John Wiley, Chichester

Conrad R 1996: Soil microorganisms as controllers of atmospheric trace gases (H_2, CO, CH_4, OCS, N_2O and NO). *Microbiol*. Rev., **60**, 604–640.

Conrad R 1999: Contribution of hydrogen to methane production and control of hydrogen concentrations in methanogenic soils and sediments. *FEMS Microbiology Ecology*, **28**, 193-202.

Conrad R 2007: Microbial Ecology of methanogens and methanotrophs. *Adv. Agron*., **96**, 1–63.

Conrad R, Klose M 1999: Anaerobic conversion of carbon dioxide to methane, acetate and propionate on washed roots. *FEMS Microbiol. Ecol.*, **30**, 147-155.

Conrad R, Klose M, Claus P, Enrich P A 2010: Methanogenic pathway, 13C isotope fractionation, and archaeal community composition in the sediment of two clear water lakes of Amazonia. *Limnol. Oceanogr.*, **55**, 689–702.

Davidson EA, Keller M, Erickson HE, Verchot LV, Veldkamp E 2000: Testing a conceptual model of soil emissions of nitrous and nitric oxides. *Bioscience*, **50**, 667–680.

De Bont JAM, Lee KK, Bouidin DF 1978: Bacterial oxidation and emission of methane in rice paddies. *Ecol. Bull*., **26**, 91-96.

De Datta SK 1981: Principles and practices of rice production. Wiley, New York. 640p.

Delgado JA, Groffman PM Nearing MA Goddard T, Reicosky D, Lal R, Kitchen NR, Rice CW, Towery D, Salon P 2011: Conservation practices to mitigate and adapt to climate change. *J. Soil Water Conserv.*, **66**, 118A-129A.

Delicato DMS 1996: Physical-Chemical Properties and Sorption Characteristics of Peat. Ph.D. thesis, School of Chemical Sciences, Dublin City University.

Dinsmore KJ, Skiba UM, Billett MF, Rees RM 2009: Effect of water table on greenhouse gas emissions from peatland mesocosms. *Plant Soil.*, **318**, 229–242.

Dobbie KE, Mc Taggart IP, Smith KA 1999: Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. *J. Geophys. Res.*, **104**, 26, 891–26,899.

Dobermann A, Witt C, Dawe D 2004: Increasing productivity of intensive rice systems through site specific nutrient management. Science Publishers, Inc., Enfield, NH, USA and International Rice Research Institute (IRRI).

Donato DC, Kauffman JB, Murdiyarso D, Kurnianto S, Stidham M, Kanninen M 2011: Mangroves among the most carbon rich forests in the tropics. *Nat. Geosci.*, **4**, 293–297,

Doorenbos J, Kassam AH 1986: Yield response of water. FAO Irrigation and Drainage Paper 33. FAO, Rome, Italy.

Dunfield P, Knowles R, Dumont R, Moore T 1993: Methane production and consumption in temperate and subarctic peat soils: response to temperature and pH. *Soil Biol. Biochem.*, **25**, 321–326.

Ehhalt DH, Schmidt U 1978: Sources and sinks of atmospheric methane. *Pure Appl. Geophys.*, **116**, 452–464.

Fageria NK, Moreira A, Coelho AM 2011: Yield and yield components of upland rice as influenced by nitrogen sources. *J. Plant Nutrition*, **34**, 361–370

FAO 2013: FAO statistical yearbook, world food and agriculture. Food and Agriculture Organization of the United Nations, Rome. 307p.

Feng J, Chen C, Zhang Y, Song Z, Deng A, Zheng C, Zhang W 2013: Impacts of cropping practices on yield-scaled greenhouse gas emissions from rice fields in China: a meta-analysis. *Agric. Ecosyst. Environ.*, **164**, 220–228.

Firmansyah MA, Nugroho WA, Mokhtar MS 2013: Sustainable peatland management: A case study of rubber plantation and intter-cropping crop in Jabiren, Pulang Pisau, Central Kalimantan. In proceeding of sustainable peatland management. pp 233–244. Indonesian Agency for Agriculture Research and Development, Bogor, Indonesia. (In Bahasa Indonesia).

Freney JR, Denmead OT 1992: Factors controlling ammonia and nitrous oxide emissions from flooded rice fields. *Ecological Bulletin*, **42**, 188-194.

Funk DW, Pullman ER, Peterson KM, Grill PM, Billings WD 1994: Influence of water table on carbon dioxide, carbon monoxide, and methane fluxes from taiga bog microcosms. *Glob. Biogeochem. Cycles*, **8**, 271–278.

Furukawa Y, Inubushi K 2002: Feasible suppression technique of methane emission from paddy soil by iron amendment. *Nutr. Cycling Agroecosyst.*, **64**, 193–201.

Furukawa Y, Inubushi K 2004: Evaluation of slag application to decrease methane emission from paddy soil and fate of iron. *Soil Sci. Plant Nutr*., **50**, 1029–1036.

Furukawa Y, Inubushi K, Ali M, Itang AM, Tsuruta H 2005: Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands. *Nutr. Cycling Agroecosyst.*, **71**, 81–91.

Ghosh S, Majumdar D, Jain MC 2003: Methane and nitrous oxide emissions from an irrigated rice of North India. *Chemosphere,* **51**, 181–195

Glaser B, Lehmann J, Zech W 2002: Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal – a review. *Biol. Fertil. Soils,* **35**, 219–230

Gleick PH 1993: Water in crisis: A guide to the world's fresh water resources. New York, N.Y. (USA): Oxford University Press.

Gorham E 1991: Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecol. Appl*., **1**, 182–195.

Gorham E, Janssens JA, Glaser PH 2003: Rates of peat accumulation during the postglacial period in 32 sites from Alaska to Newfoundland, with special emphasis on northern Minnesota. *Can. J. Bot.*, **81**, 429–438.

Granli T, Bockman O C 1994: Nitrous oxide from agricultural. *Norwegian Journal of Agric. Scie*., **12**, 7–128.

Hadi A, Haridi M, Inubushi K, Purnomo E, Razie F, Tsuruta H 2001: Effect of land-use change in tropical peat soil on microbial population and emission of greenhouse gases. *Microbes environ*., **2**, 79–86.

Hadi A, Inubushi K, Yagi K 2010: Effect of water management on greenhouse gas emissions and microbial properties from paddy fields of Indonesia and Japan. *Paddy Water Environ*., **8**, 319-324

Hadi A, Fatah L, Syaifuddin, Abdullah, Affandi DN, Bakar RA and Inubushi K 2012: Greenhouse Gas Emissions from Peat Soils Cultivated to Rice Field, Oil Palm and Vegetable. *J. Trop. Soils,* **17**, 105-114

Hadi A, Inubushi K, Furukawa Y, PurnomoE, Rasmadi M, Tsuruta H 2005: Greenhouse gas emissions from tropical peatlands of Kalimantan, Indonesia. *Nutr. Cycling Agroecosyst*., **71**: 73–80.

Hatta S 1967: Water consumption in paddy field and water saving rice culture in the tropical zone. *Jpn. Trop. Agric*., **11**, 106-112.

Hikmatullah, Hidayat H, Suryana U 2013: Detail mapping of peat soil at demonstration plot to support carbon emissions experiment in Jabiren, Central Kalimantan Demonstration. In proceeding of sustainable peatland management. pp 129–142. Indonesian Agency for Agriculture Research and Development, Bogor, Indonesia. (In Bahasa Indonesia).

Hirano T, Segah H, Harada T, Limin S, June T, Hirata R, Osaki M 2007: Carbon dioxide balance of a tropical peat swamp forest in Kalimantan, Indonesia. *Global Change Biol.*, **13**, 412–425.

Hobbs PR 2007: Conservation agriculture: what is it and why is it important for future sustainable food production? *J. Agric. Sci.*, **145**, 127-138

Hooijer A, Silvius M, Wosten H, Page S 2006: PEAT-CO_2, Assessment of CO_2 emissions from drained peatlands in SE Asia. Report Q3943. Delft Hydraulics, Delft, the Netherlands.

Hou AX, Chen GX, Wang ZP, Cleemput OV, Patrick Jr WH 2000: Methane and nitrous oxide emissions from a rice field in relation to soil redox and microbiological processes. *Soil Sci. Soc. Am. J.*, **64**, 280-286.

http://blogs.helsinki.fi/jyjauhia/peat-in-agriculture-and-forestry downloaded on September 2015

http://geology.usgs.gov/postdoc/profiles/moseman/ downloaded on September 2015

https://www1.ethz.ch/ibp/research/environmentalmicrobiology/research/Wetlands downloaded on September 2015

Huang B, Yu K, Gambrell RP 2009: Effects of ferric iron reduction and regeneration on nitrous oxide and methane emissions in a rice soil. *Chemosphere*, **74**, 481–486.

Hue NV 1992. Correcting soil acidity of a highly weathered Ultisol with chicken manure and sewage sludge. *Commun. Soil Sci. Plant Anal.*, **23**, 241–264.

Hutchinson JJ, Campbell CA, Desjardins RL 2007: Some perspectives on carbon sequestration in agriculture. *Agr. Meteorol.*. **142**, 228–302.

IAEA-International Atomic Energy Agency 1992: Manual on measurement of methane and nitrous oxide emission from agricultural. pp. 91. IAEA-TECDOC-674 Vienna: IAEA.

ICCTF (Indonesia Climate Change Trust Fund) 2011: research and development of technology on sustainable peatland management to enhance carbon sequestration and greenhouse gas mitigation. Final presentation on greenhouse gas emission. (In Bahasa Indonesia)

Immirzi CP, Maltby E, Clymo RS 1992: The global status of peatlands and their role in carbon cycling. A report for the Friends of the Earth by the Wetlands Ecosystems Research Group, Department of Geography, University of Exeter. Friends of the Earth, London.

Inubushi K, Furukawa Y, Hadi A, Purnomo E, Tsuruta H 2003: Seasonal changes of CO_2, CH_4, and N_2O fluxes in relation to land-use change in tropical peatlands located in coastal area of South Kalimantan. *Chemosphere*, **52**, 603–608.

Inubushi K, Hori K, Matsumoto S, Umebayashi M Wada H 1989: Methane flux through rice plant to the atmosphere, *Japanese J. Soil Sci. Plant Nutr.*, **60**, 318-324. (in Japanese with English summary).

Inubushi K, Umebayashi M, Wada H 1990: Methane emission from paddy fields. *In* Transactions of 14th International Congress of Soil Science vol. 2, p249-254, *Int. Soc. Soil Sci.*, Kyoto.

Inubushi K, Wada H, Takai Y 1984: Easily decomposable organic matter in paddy soil. Relationship between reduction process and organic matter decomposition. *Soil Sci. Plant Nutr.*, **30**, 189–198.

IPCC 1992: The supplementary report to IPCC scientific assessment [Houghton JT, Callander BA, Varney SK (Eds)]. Cambridge University Press, Cambridge

IPCC 1996. Climate Change 1995, Cambridge University Press, Cambridge.

IPCC 2007a. Agriculture. *In*: Metz B, Davidson OR, Bosch PR *et al.* (Eds.), Climate Change 2007: Mitigation, Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK/New York, pp. 498–540.

IPCC 2007b. Changes in atmospheric constituents and in radiative forcing. *In*: Solomon S, Qin D, Manning M *et al*. (Eds.), Climate Change 2007: The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK/New York, pp. 498–540.

IRRI 2009: Consumption and calorie supply: Rough rice consumption, by country and geographical region- USDA. (http://beta.irri.org/solutions/index.php?option=com_content&task=view&id=250) downloaded on September 2015

ISRI (Indonesian Soil Research Institute) 2006: Map of distribution of soil in Indonesia, Bogor.

Ito S 2015: World Food Statistics and Graphics. http://worldfood.apoinet.or.jp. Downloaded on September 2015

Jaafar MN, Saad P, Vel Arjunan SN 2000: A computational model for water use efficiency in rice production. *Jurnal Teknologi Maklumat*, **12**, 1-13.

Jacinthe PA, Lal R, Kimble JM 2002: Carbon budget and seasonal carbon dioxide emission from a central Ohio Luvisol as influenced by wheat residue amendment. *Soil Tillage Res*., **67**, 147–157.

Jackel U, Russo S, Schnell S 2005: Enhanced iron reduction by iron supplement: A strategy to reduce methane emission from paddies. *Soil Biol. Biochemist.*, **37**, 2150–2154.

Jaenicke J, Rieley JO, Mott C, Kimman P, Siegert F 2008: Determination of the amount of carbon stored in Indonesian peatlands. *Geoderma*, **147**, 151–158.

Jakobsen P, Patrick Jr WH, Williams BG 1981: Sulfide and methane formation in soils and sediments. *Soil Sci.*, **132**, 279–287.

Jali D 2004: Nitrogen mineralization in the tropical peat swamps. *In*: Proceedings of the 12th International Peat Congress, Tampere 6-11 June 2004. Edited by: P¨aiv¨anen, J, pp 644–652.

Jauhiainen J, Silvennoinen H 2012: Diffusion GHG fluxes at tropical peatland drainage canal water surfaces. *Suoseura*, **63**, 93–105

Johnson JMF, Franzluebbers AJ, Weyers SL, Reicosky DC 2007: Agricultural opportunities to mitigate greenhouse gas emissions. *Environmental Pollution.*, **150**, 107–124.

Johnson-Beebout SE, Angeles OR, Alberto MCR, Buresh RJ 2009: Simultaneous minimization of nitrous oxide and methane emission from rice paddy soils is improbable due to redox potential changes with depth in a greenhouse experiment without plants. *Geoderma*, **149**, 45–53.

Jones DL, Murphy DV, Khalid M, Ahmad W, Edwards-Jones G, DeLuca TH 2011: Short-term biochar-induced increase in soil CO_2 release is both biotically and abiotically mediated. *Soil Biol. Biochem.*, **43**, 1723–1731.

Jungkunst HF, Fiedler S, Stahr K 2004: N_2O emissions of a mature Norway spruce (Picea abies) stand in the Black Forest (SW Germany) as differentiated by the soil pattern. *J. Geophys. Res.*, **109**, D07302.

Jungkunst HF, Flessa H, Scherber C, Fiedler S 2008: Groundwater level controls CO_2, N_2O and CH_4 fluxes of three different hydromorphic soil types of a temperate forest ecosystem. *Soil Biol. Biochem.*, **40**, 2047–2054.

Kampschreur MJ, Kleerebezem R, De Vet WWJM, Van Loosdrecht MCM 2011: Reduced iron induced nitric oxide and nitrous oxide emission. *Water research,* **45**, 5945–5952.

Kang G, Cai Z, Feng X 2002. Importance of water regime during the non-rice growing period in winter in regional variation of CH_4 emissions from rice fields during following rice growing period in China. *Nutr. Cycl. Agroecosyst.*, **64**, 95–100.

Karhu K, Mattila T, Bergstrom I, Regina K 2011: Biochar addition to agricultural soil increased CH_4 uptake and water holding capacity: results from a short-term pilot field study. *Agr Ecosyst. Environ.,* **140**, 309–313.

Katayanagi N, Furukawa Y, Fumoto T, Hosen Y 2012: Validation of the DNDC-Rice model by using CH_4 and N_2O flux data from rice cultivated in pots under alternate wetting and drying irrigation management. *Soil Sci. Plant Nutr.*, **58**, 360—372.

Kauffman JB, Heider C, Cole TG, Dwire KA, Donato DC 2011: Ecosystem carbon stocks of Micronesian mangrove forests. *Wetlands*, **31**, 343–352.

Khalil K, Mary B, Renault P 2004: Nitrous oxide production by nitrification and denitrification in soil aggregates as affected by O_2 concentration. *Soil Biol. Biochem.*, **36**, 687–699.

Khosa MK, Sidhu BS, Benbi DK 2010: Effect of organic materials and rice cultivars on methane emission from rice field. *J. Environ. Biol*., **31**, 281-285.

Kimura M 1992: Methane emission from paddy soils in Japan and Thailand in Proceeding of an International Workshop Organized in the Framework of the Dutch National Research Programme on Global Air Pollution and Climate Change (Wageningen 24- 27 August 1992) edited by Batjesa NH, Bridges EM, WISE Rep. 2 , pp. 43-79, Int. Soil Ref. and Inf. Cent., Wageningen Netherlands

Kimura M 1992: Methane emissions from paddy soils in Japan and Thailand. *In* proceeding of an international workshop on Global Air Pollution and Climate Change. Wageningen 24- 27 August 1992) edited by Batjes NH, Bridges EM. WISE Rep.2. pp. 43-79. Wageningen, Netherlands.

King GM, Roslev P, Skovgaard H 1990: Distribution and rate of methane oxidation in sediments of the Florida Everglades. *Appl. Environ. Microbiol.*, **56**, 2902–2911.

Klemedtsson L, Svensson BH, Rosswall T 1988: Relationships between soil moisture content and nitrous oxide production. *Biol. Fertil. Soils.*, **6**, 106-111.

Koma, Y.S. and Sinv, S. (2003). An assessment of ecological system of rice intensification (SRI) in Combodian wet season. CEDAC field document, Pheom penti, CEDAC mimeo.

Kreye C, Bouman BAM, Reversat G, Fernandez L, Vera Cruz L, Elazegui F, Faronilo JE, Liorca L 2009. Biotic and abiotic causes of yield failure in tropical aerobic rice. *Field Crops Res*., **112**, 97–106.

Kroeze C, Mosier A, Bouwman L 1999: Closing the global N_2O budget: A retrospective analysis 1500–1994. *Global Biogeochem. Cycles*, **13**, 1–8.

Kruger M, Eller G, Conrad R, Frenzel P 2002: Seasonal variation in pathways of CH_4 production and in CH_4 oxidation in rice fields determined by stable carbon isotopes and specific inhibitors. *Glob. Change Biol.*, **8**, 265–280.

Kürschner E, Henschel C, Hildebrandt T, Jülich E, Leineweber M, Paul C 2010: Water saving in rice production–dissemination, adoption and short term impacts of alternate wetting and drying (AWD) in Bangladesh. Humboldt Universität zu Berlin. 96p.

Lafleur PM, Moore TR, Roulet NT Frolking S 2005: Ecosystem respiration in a cool temperate bog depends on peat temperature but not water table. *Ecosystems,* **8**, 619–629.

Langner A, Miettinen J, Siegert F 2007: Land cover change 2002–2005 and the role of fire derived from MODIS imagery. *Global Change Biol*., **13**, 1–12.

Lantin RS, Aduna JB, Javellana AMJ 1995: Methane measurements in rice fields. Instruction manual and methodologies, maintenance and troubleshooting guide. A joint undertaking by: International Rice Research Institute (IRRI), United State Environmental Protection Agency (US-EPA) and United Nation Development Program (UNDP).

Las I, Surmaini W 2010: Agricultural development technologies to reduce global warming. Paper on One Day Seminar for Alumni Gathering of Agricultural Faculty of Lambung Mangkurat University, Banjarbaru, March 11, 2010 (in Bahasa Indonesia).

Lee CH, Kim SY, Villamil MB, Pramanik P, Hong CO, Kim PJ 2012: Different response of silicate fertilizer having electron acceptors on methane emission in rice paddy soil under green manuring. *Biol. Fertil. Soils,* **48**, 435–442.

Lehmann J, Rillig MC, Thies J, Masiello CA, Hockaday WC, Crowley D 2011: Biochar effects on soil biota—a review. *Soil Biol. Biochem*., **43**, 1812–1836.

Lehmann J, Sohi S 2008: Comment on “fire-derived charcoal causes loss of forest humus”. *Science*, 321: 1295

Levitt J 1980: Responses of plants to environmental stresses. Second eds. New York: Academic Press.

Li XL, Yuan WP, Xu H, Cai ZC, Yagi K 2011: Effect of timing and duration of midseason aeration on CH_4 and N_2O emissions from irrigated lowland rice paddies in China. *Nutr. Cycl. Agroecosyst*., **91**, 293–305.

Lin M, You C 1989: Root exudates of rice (Oryza sativa L) and its interaction with Alcaligenes faecalis. *Sci. Agric. Sinica*, **22**, 6–12.

Linn DM, Doran JW 1984: Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and non-tilled soils. *Soil Sci. Soc. Am. J.*, **48**, 1267–1272.

Liu S, Zhang L, Liu Q, Zou J 2012: Fe(III) fertilization mitigating net global warming potential and greenhouse gas intensity in paddy rice-wheat rotation systems in China. *Environ. Pollution,* **164**, 73–80.

Liu SW, Qin YM, Zou JW, Liu QH 2010: Effects of water regime during rice-growing season on annual direct N_2O emission in a paddy rice-winter wheat rotation system in southeast China. *Sci. Total Environ.,* **408**, 906–913.

Liu Y, Wan K, Tao Y, Li Z, Zhang G, Li S, Chen F 2013: Carbon dioxide flux from rice paddy soils in Central China: Effects of intermittent flooding and draining cycles. www.plosone.org

Liu YX, Yang M, Wu YM, Wang HL, Chen YX, Wu WX 2011: Reducing CH_4 and CO_2 emissions from waterlogged paddy soil with biochar. *J. Soils Sediments*, **11**, 930–939.

Lobb DA 1997: Management and conservation practices for vegetable production on peat soils. pp 1–58. University of Moncton, Canada.

Lovley DR, Phillips EJP 1988: Manganese inhibition of microbial iron reduction in anaerobic sediments. *Geomicrobiol.,* **6**, 145–155.

Ma J, Ma E, Xu H, Yagi K, Cai Z 2009: Wheat straw management affects CH_4 and N_2O emissions from rice fields. *Soil Biol. Biochem.,* **41**, 1022–1028.

Ma JF, Takahashi E 2002: Soil, Fertilizer, and Plant Silicon Research in Japan. Pp 107–180. Elsevier Science, Amsterdam.

Maas A 1996: A note on the formation of peat deposits in Indonesia. *In*: Maltby E, Immirzi CP, Safford RJ. (eds), Tropical Lowland Peatlands of Southeast Asia. Proceedings of a Workshop on Integrated Planning and Management of Tropical Lowland Peatlands. IUCN, Gland, Switzerland.

Maclean Jl, Dawe DC, Hardy B, Hettel GP 2002: Rice almanac: Source Book for the Most Important Economic Activity on Earth. International Rice Research Institute, Philippines, pp 85.

Maltby E, Proctor MCF 1996: Peatlands: their nature and role in the biosphere. *In*: Lappalainen E. (ed), Global Peat Resources. International Peat Society and Geological Survey of Finland, Jyska Finland.

Masscheleyn PH, Delaune RD, Patrick Jr WH 1993: Methane and nitrous oxide emission from laboratory measurements of rice soil suspension: effect of soil oxidation-reduction status. *Chemosphere,* **26**, 251–260.

Matson PA, Parton WJ, Power AG, Swift MJ 1997: Agricultural intensification and ecosystem properties. *Science*, 277, 25 July 1997. www.sciencemag.org.

Melling L, Hatano R, Goh KJ 2005: Nitrous oxide emissions from three ecosystems in tropical peatland of Sarawak, Malaysia. *Soil Sci. Plant Nutr.,* **53**, 792–805.

Melling L, Hatano R, Goh KJ 2005: Soil CO_2 flux from three ecosystems in tropical peatland of Sarawak, Malaysia. *Tellus,* ***57B***, 1–11.

Minamikawa K, Nishimura S, Sawamoto T, Nakajima Y, Yagi K 2010: Annual emissions of dissolved CO_2, CH_4, and N_2O in the subsurface drainage from three cropping systems. *Global Change Biol.*, **16**, 796–809.

Minamikawa K, Sakai N 2005: The practical use of water management based on soil redox potential for decreasing methane emission from a paddy field in Japan. *Agriculture, Ecosyst. Environ.*, **116**, 181–188.

Mitra S, Wassmann R, Jain MC, Pathak H 2002: Properties of rice soils affecting methane production potentials: 1. Temporal patterns and diagnostic procedures. *Nutr. Cycl. Agroecosyst.*, **64**, 169–182.

Miyata A, Leuning R, Denmead OT, Kim J, Harazono Y 2000: Carbon dioxide and methane fluxes from an intermittently flooded paddy field. *Agricultural Forest Meteorol.*, **102**, 287–303.

Monteny GJ, Bannink A, Chadwick D 2006: Greenhouse gas abatement strategies for animal husbandry. *Agric. Ecosyst. Environ.*, **112**, 163–170.

Moore TR, Dalva M 1993: The influence of temperature and water-table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. *J. Soil Sci.*, **44**, 651–664.

Moore TR, Knowles R 1988: The influence of water table levels on methane and carbon dioxide emissions from peatland soils. *Can. J. Soil Sci.*, **69**, 33–38.

Mosier A, Kroeze C 2000. Potential impact on the global atmospheric N_2O budget of the increased nitrogen input required to meet future global food demands. *Chemosphere—Global Change Science*, **2**, 465–473.

Mosier A, Kroeze C, Nevison O, Oenema S, Seitzinger S, Van Cleemput O 1998: Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle - OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology. *Nutr. Cycling Agroecosyst.*, **52**, 225–248.

Mosier A, Schimel D, Valentine D, Bronson K, Parton W 1991: Methane and nitrous oxide flux in native, fertilized and cultivated grassland. *Nature*, **350**, 330-332.

Mosier AR, Duxbury JM, Freney JR, Heinemeyer O, Minami K, Johnson DE 1998: Mitigating agricultural emissions of methane. *Clim. Change*, **40**, 39–80.

Mosier AR, Halvorson AD, Peterson GA, Robertson GP, Sherrod L 2005: Measurement of net global warming potential in three agroecosystems. *Nutr. Cycl. Agroecosyst.*, **72**, 67–76.

Mukherjee A, Lal R 2013: Biochar impacts on soil physical properties and greenhouse gas emissions. *Agronomy*, **3**, 313–339.

Murdiyarso D, Hergoualc'h K, Verchot LV 2010: Opportunities for reducing greenhouse gas emissions in tropical peatlands. *P. Natl. Acad. Sci. USA*, **107**, 19655–19660

Neue HU 1985: Organic matter dynamics in wetland soil. In: wetland soils: Characterization, Classification, and utilization. Proceedings of a workshop held 26 March to 5 April 1984 under the joint sponsorship of The International Rice Research Institute.

Neue HU 1997: Fluxes of methane from rice fields and potential mitigation. *Soil Use Management*, **13**, 258–267.

Neue HU, Latin RS, Wassmann R, Aduna JB, Alberto CR, Andales MJF 1994: Methane emission from rice soils of the Philippines. In: CH_4 and N_2O: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources (eds Minami K, Mosier A, Sass R), pp. 55-63, Yokendo Publishers, Tokyo

Neue HU, Roger PA 1993: Rice agriculture: factors controlling methane emissions. *Global Environ. Change*, **13**, 254–298.

Neue HU, Sass RL 1994: Trace gas emission from rice fields. *IG Activities*, **17**, 3–11.

Neue HU, Sass RL 1998: The budget of methane from rice fields. *IGACtivities Newsletter*, **12**, 3–11.

Nielsen JL, Nielsen PH 1998: Microbial nitrate-dependent oxidation of ferrous iron in activated sludge. *Environ. Sci. Technol.*, **32**, 3556–3561.

Nieveen JP, Campbell DI, Schipper LA and Blair IJ 2005: Carbon exchange of grazed pasture on a drained peat soil. *Glob. Chang. Biol.*, **11**, 607–618.

Nishio T, Fujimoto T 1990: Kinetics of nitrification of various amounts of ammonium added to soils. *Soil Biol. Biochem.*, **22**, 51–55.

Noor M 2012: Indigenous knowledge on peatland management. Proceeding of sustainability of peatland management. Indonesian Center for Agricultural Land Resources Research and Development (ICALRRD), Bogor, Indonesia. p 159. (In Bahasa Indonesia)

Noubactep C 2011: On the mechanism of microbe inactivation by metallic iron. *J.Hazard. Mater*, **198**, 383–386.

Nykanen H, Alm J, Silvola J, Tolonen K, Martikainen PJ 1998: Methane fluxes on boreal peatlands of different fertility and the effect of long-term experimental lowering of the water table on flux rates. *Global Biogeochem. Cycles*, **12**, 53–69.

Olaleye, A.O, Ogunkunle, A.O, Sharawat, K.L and Singh, B.N (2001). Characterisation of some benchmark wetlands pedons in Nigeria for Rice cultivation. *Communication in soil science and plant analysis* 32(1&2):189-208.

Oliver MMH, Talukder MSU, Ahmed M 2008: Alternate wetting and drying irrigation for rice cultivation. *J. Bangladesh Agril. Univ*, **6**, 409–414.

Olivier JGJ, Bouwman AF, Van der Hoek KW, Berdowski JJM 1998: Global air emission inventories for anthropogenic sources of NO_x, NH_3 and N_2O in 1990. *Environ. Pollut.*, **102**, 135–48.

Osaki M, Shinano T, Matsumoto M, Zheng T, Tadano T 1997. A root-shoot interaction hypothesis for high productivity of field crops. *Soil Sci. Plant Nutr.*, **43**, 1079–1084.

Ottow JCG 1981: Mechanisms of Bacterial Iron-Reduction in Flooded Soils. In: Proceedings of Symposium on Paddy Soils. Institute of Soil Science, Academia Sinica

Ou SH 1979: Rice plant diseases. *In* AVI Publishing Co. Rice: production and utilization. Westport, Connecticut. Pages 235-259

Page SE, Rieley JO, Banks CJ 2011: Global and regional importance of the tropical peatland carbon pool. *Glob. Chang. Biol.*, **17**, 798–818.

Page SE, Rieley JO, Shotyk W, Weiss D 1999: Interdependence of peat and vegetation in a tropical peat swamp forest. Proceedings of the Royal Society, B 354, 1–13.

Page SE, Siegert F, Rieley JO, Boehm H-DV, Jaya A, Limin S 2002: The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, **420**, 61–65.

Page SE, W¨ust, RAJ, Weiss D, Rieley JO, Shotyk W, Limin SH 2004: A record of Late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): implications for past, present and future carbon dynamics, *J. Quaternary Sci.*, **19**, 625–635.

Pan GX, Li LQ, Wu LS, Zhang XH 2004: Storage and sequestration potential of topsoil organic carbon in China's paddy soils. *Global Change Biol.*, **10**, 79–92.

Patel D, Das A, Munda G, Ghosh P, Bordoloi JS, Kumar M 2010: Evaluation of yield and physiological attributes of high-yielding rice varieties under aerobic and flood-irrigated management practices in mid-hills ecosystem. *Agric. Water Manag.*, **97**, 1269-1276.

Pathak H 1999: Emissions of nitrous oxide from soil. *Current science*, **77**, 359–369

Paustian K, Cole CV, Dieter S, Sampson N 1998: CO_2 mitigation by agriculture: an overview. *Climatic Change,* **40**, 135–162.

Peng S, Bouman BAM 2007: Prospects for genetic improvement to increase lowland rice yields with less water and nitrogen. In: Spiertz JHJ, Struik PC, van Laar HH, eds. Scale and complexity in plant systems research: gene-plant-crop relations. pp. 251–266. New York: Springer.

Peng S, Hou H, Xu J, Mao Z, Abudu S, Luo Y 2011: Nitrous oxide emissions from paddy fields under different water managements in southeast China. *Paddy Water Environ.,* **9**,403–411.

Philippot L, Hallin S 2011: Towards food, feed and energy crops mitigating climate change. *Trends in Plant Science,* **16**, 476-480

Ponnamperuma FN 1981: Some aspects of the physical chemistry of paddy soils. In: Sinica, A. (Ed.), Proceedings of Symposium on Paddy Soil, pp. 59–94. Science Press-Springer, Beijing/Berlin.

Ponnamporuma FN 1972: The chemistry of submerged soils. *Adv. Agron.*, **24**, 29–96

Presidential regulation no 61 year 2011 of the Republic of Indonesia on the national action plan for greenhouse gas emissions reduction.

Raimbauit M, Rinaudo G, Garcia JL, Borean M 1977: A device to study metabolic gases in the rice rhizosphere. *Soil Biol. Biochem.*, **9**, 193-196.

Reddy KR, Patrick WH Jr, Lindau CW 1989: Nitrification-denitrification at the plant root sediment interface in wetlands. *Limnol. Oceanogr.,* **34**, 1004–1013.

Regina K, Nykanen H, Silvola J, Martikainen PJ 1996: Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. *Biogeochem.,* **35**, 401–418.

Rejesus RM, Palis FG, Rodriguez DGP, Lampayan RM, Bouman BAM 2011: Impact of the alternate wetting and drying (AWD) water-saving irrigation technique: Evidence from rice producers in the Philippines. *Food Policy,* **36**, 280–288.

Ridlo R. 1997. CO_2 emissions at peat reclamation for mega rice project in Central Kalimantan. *J. Alami.*, **2**, 57-58 (in Bahasa Indonesia).

Rieley JO, Ahmad-Shah AA, Brady M, 1996: The extent and nature of tropical peat swamps. In: Maltby E, Immirzi CP, Safford RJ (Eds.): Tropical Lowland Peatlands of Southeast Asia. Proceedings of a Workshop on Integrated Planning and Management of Tropical Lowland Peatlands, Cisarua, Indonesia. IUCN, Gland, Switzerland. 294 pp.

Rieley JO, Page SE (Eds.) 2005: Wise Use of Tropical Peatlands: Focus on Southeast Asia. Alterra, Netherlands. At www.restorpeat.alterra.wur.nl.

Ritung S, Wahyunto, Sukarman 2011: Peat land map of Indonesia, scale 1:250.000. Indonesian Center for Agro-climate Research and Development, Indonesian Agency for Agriculture Research and Development, Ministry of Agriculture, Jakarta, Indonesia.

Robertson GP, Paul EA, Harwood RR 2000: Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *Science*, **289**, 1922–1925.

Roulet N, Moore TR, Bubier J, Lafleur P 1992: Northern fens: methane flux and climatic change. *Tellus* (B), **44**, 100–105.

Roy R, Klüber HD, Conrad RF 1997: Early initiation of methane production in anoxic soil despite the presence of oxidants. *FEMS Microbiol. Ecol.*, **12**, 311–320.

Rusera R, Flessa H, Russow R, Schmidt G, Buegger F, Muncha JC 2006: Emission of N_2O, N_2 and CO_2 from soil fertilized with nitrate: effect of compaction, soil moisture and rewetting. *Soil Biol. Biochem.*, **38**, 263–274.

Sabiham S 1988: Studies on peat in the coastal plains of Sumatra and Borneo: I. Physiography and geomorphology of the coastal plains. *Southeast Asian Studies*, **26**, 308-335.

Saharjo BH 2011: Carbon baseline as limiting factor in managing environmental sound activities in peatland for reducing greenhouse gas emission. *Biodiversitas*, **3**, 182–186.

Sahrawat KL 2000: Elemental composition of the rice plant as affected by iron toxicity under field conditions. *Commun. Soil Sci. Plant Anal.*, **132**, 2819–2827.

Sakata R, Shimada S, Arai H, Yoshioka N, Yoshioka R, Aoki H, Kimoto N, Sakamoto A, Melling L, Inubushi K 2015: Effect of soil types and nitrogen fertilizer on nitrous oxide and carbon dioxide emissions in oil palm plantations. *Soil Sci. Plant Nutr.*, **61**, 48–60

Sakata R, Shimada S, Arai H, Yoshioka N, Yoshioka R, Kimoto N, Sakamoto A, Melling L, Inubushi K 2015: Effect of soil types and nitrogen fertilizer on nitrous oxide and carbon dioxide emissions in oil palm plantations. *Soil Sci. Plant Nutr.*, **61**, 48-60.

SAS Institute 2003: *System for Windows Release 9.1*. SAS Institute, Carry, NC.

Sass RL, Fisher FM, Wang YB 1992: Methane emission from rice fields: the effect of floodwater management. *Global Biochem. Cycles*, **6**, 249-262

Sato S, Uphoff N 2007: A review of on-farm evaluation of system of rice intensification (SRI) methods in eastern Indonesia. *CAB Reviews*: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources. Wallingford: Commonwealth Agricultural Bureau International.

Schutz H, Seiler W, Conrad R 1989: Processes involved in formation and emission of methane in rice paddies. *Biogeochem.*, **7**, 33–53.

Shang QY, Yang XX, Gao CM, Wu PP, Liu JJ, Xu YC, Shen QR, Zou JW, Guo SW 2010: Net annual global warming potential and greenhouse gas intensity in Chinese double rice-cropping systems: a 3-year field measurement in long-term fertilizer experiments. *Global Change Biol*., **17**, 2196–2210.

Sharma PK 1989: Effect of Period Moisture Stress on Water-use Efficiency in Wetland Rice. *Oryza,* **26**, 252-257.

Siegert F, Ruecker G, Hinrichs A, Hoffmann AA 2001: Increased damage from fires in logged forests during droughts caused by El Niño. *Nature*, **414**, 437–440.

Sigren LK, Byrd GT, Fisher FM, Sass RL 1997: Comparison of soil acetate concentration and methane production, transport and emission in two rice cultivars. *Global Biochem. Cycles,* **11**, 1–14.

Silva CC, Guido ML, Ceballos JM, Marsch R, Dendooven L 2008: Production of carbon dioxide and nitrous oxide in alkaline saline soil of Texcoco at different water contents amended with urea: A laboratory study. *Soil Biol. Biochem.,* **40**, 1813–1822.

Silvola J, Alm J, Ahlholm U, Nykanen H, Martikainen PJ 1996: CO_2 fluxes from peat in boreal mires under varying temperature and moisture conditions. *J. Ecology*, **84**, 219–228.

Singh BP, Hatton BJ, Singh B, Cowie AL, Kathuria A 2010: Influence of biochars on nitrous oxide emission and nitrogen leaching from two contrasting soils. *J. Environ. Qual.,* **39**, 1224–1235.

Singh CB, Aujla TS, Sandhu BS, Khera KL 1996: Effects of transplanting data and irrigation regime on growth, yield and water use in rice (Oryza sativa) in northern India. *Indian J. Agricultural Sciences,* **66**, 137-141.

Singh S, Singh JS, Kashyap AK 1999: Methane flux from irrigated rice fields in relation to crop growth and N-fertilization. *Soil Biol. Biochem.,* **31**, 1219–1228.

Singla A, Inubushi K 2013: Effect of slag-type fertilizers on N_2O flux from komatsuna vegetated soil and CH_4 flux from paddy vegetated soil. *Paddy Water Environ*., **13**, 43–50.

Singla A, Inubushi K 2014: Effect of biochar on CH_4 and N_2O emission from soils vegetated with paddy. *Paddy Water Environ.,* **12**, 239–243.

Skopp J, Jawson MD, Doran JW 1990: Steady-state aerobic microbial activity as a function of soil-water content. *Soil Sci. Soc. Am. J.,* **54**, 1619-1625.

Smith CJ, Patrick WH 1983: Nitrous oxide emission as affected by alternate anaerobic and aerobic conditions from soil suspensions enriched with ammonium sulfate. *Soil Biol. Biochem.,* **15**, 693–697.

Smith JL, Collins HP, Bailey VL 2010: The effect of young biochar on soil respiration. *Soil Biol. Biochem.*, **42**, 2345–2347.

SNC 2010: Indonesia second national communication under the United Nations Framework Convention on Climate Change (UNFCCC). Ministry of Environment, Republic of Indonesia, Jakarta.

Statistic Indonesia (BPS) 2014: Table of harvested area-productivity-production paddy in all provinces. http://www.bps.go.id/eng/tnmn_pgn.php?kat=3&id_subyek=53&notab=0 (December, 2014).

Stepniewski W, Stepniewska Z 1998: Oxygenology as a new discipline in the environmental sciences. *Int Agrophys*., **12**, 53–55.

Stoop WA, Uphoff N, Kassam A 2002: A review of agricultural research issue raised by the System of Rice Intensification (SRI) from Madagascar: Opportunities for improving system for resource poor farmers. *Agric. Syst.*, **71**, 249-274.

Sudhir-Yadav, Humphreys E, Li T, Gill G, Kukal SS 2012: Evaluation of tradeoffs in land and water productivity of dry seeded rice as affected by irrigation schedule. *Field Crops Res*., **128**, 180–190.

Sugano A, Tsuchimoto H, Cho Cho T, Kimura M, Asakawa S 2005: Succession of methanogenic archaea in rice straw incorporated into a Japanese rice field: estimation by PCR-DGGE and sequence analyses. *Archaea*, **1**, 391–397

Susilawati HL, Setyanto P, Makarim AK, Ariani M, Ito K, Inubushi K 2015: Effects of steel slag applications on CH_4, N_2O and the yields of Indonesian rice fields: a case study during 2 consecutive rice-growing seasons at 2 sites. *Soil Sci. Plant Nutr.*, **16**, 704–718.

Tabbal DF, Lampayan R, Bhuiyan SI 1992: Water-efficient irrigation technique for rice. *In*: Maurty VVN, Koga K (eds) Soil and water engineering for paddy field management. Proceedings of the International Workshop on Soil and water engineering for paddy field management, Bangkok, Thailand, 28–30 January 1992. pp 146–159. Asian Institute of Technology, Bangkok.

Takai Y, Wada E 1990: Methane formation in waterlogged paddy soils and its controlling factors. In: Scharpenseel HW, Schomaker M, Ayoub A (eds): Soils on a Warmer Earth. Developments in Soil Science, Vol 20. pp 101–107. Elsevier.

Takakai F, Morishita T, Hashidoko Y, Darung U, Kuramochi K, Dohong S, Limin SH, Hatano R 2006: Effects of agricultural land-use change and forest fire on N_2O emission from tropical peatlands, Central Kalimantan, Indonesia. *Soil Sci. Plant Nutr*., **52**, 662–674.

Thornton PK, Herrero M 2009: The inter-linkages between rapid growth in livestock production, climate change, and the impacts on water resources, land use, and deforestation. World Bank Policy Research Working Paper, WPS 5178. World Bank, Washington, DC, USA

Tiessen H, Cuevas E, Chacon P 1994: The role of soil organic matter in sustaining soil fertility. *Nature*, **371**, 783–785.

Tuong TP, Bouman BAM 2003: Rice Production in Water-scarce Environments. *In*: *Water Productivity in Agriculture: Limits and Opportunities for Improvement* (eds J.W. Kijne, R. Barker and D. Molden). pp. 53-67. CABI Publishing, Wallingford, UK.

Tuong TP, Bouman BAM, Mortimer M 2005: More rice, less water: integrated approaches for increasing water productivity in irrigated rice-based systems in Asia. *Plant Production Science*, **8**, 231–241.

Ueda S, Go C-SU, Yoshioka T, Yoshida N, Wada E, Miyajima T, Sugimoto A, Boontanon N, Vijarnsorn P, Boonprakub S 2000: Dynamics of dissolved O_2, CO_2, CH_4, and N_2O in a tropical coastal swamp in southern Thailand. *Biogeochem.*, **49**, 191–215.

Uphoff N 2006: The System of Rice Intensification (SRI) as a methodology for reducing water requirements in irrigated rice production. Paper for International Dialogue on Rice and Water: Exploring Options for Food Security and Sustainable Environments, held at IRRI, 7-8 March 2006, Los Baños, Philippines.

US EPA 2006: Global Mitigation of Non CO_2 Greenhouse Gases (EPA Report 430-R-06–005). United States Environmental Protection Agency, Office of Atmospheric Programs (6207J), Washington, DC. Available from:. http://www.epa.gov/nonco2/econ-inv/international.html

USDA 2014: Keys to soil taxonomy. United States Department of Agriculture Natural Resources Conservation Service. Twelfth edition. 360p.

Van Cleemput O 1998: Subsoils: chemo and biological denitrification, N_2O and N_2 emissions. *Nutr. Cycling Agroecosyst*., **52**, 187–194.

Van Cleemput O, Baert L 1983: Nitrite stability influenced by iron compounds. *Soil Biol. Biochemist*., **15**, 137–140.

Van der Hoek W, Sakthivadivel R, Renshaw M, Silver J B, Birley MH, Konradsen F 2001: Alternate wet/dry irrigation in rice cultivation: A practical way to save water and control malaria and Japanese encephalitis? Research Report 47, Colombo: International Water Management Institute.

Van der Watt H, Barnard RO, Cronje IJ, Dekker J, Croft GJB, Van der Walt MM 1991: Amelioration of subsoil acidity by application of a coal-derived calcium fulvate to the soil surface. *Nature*, **350**, 146–148.

Van Nguyen N, Ferrero A 2006: Meeting the challenges of global rice production. *Paddy Water Environ.*, **4**, 1–9.

Verge´ XPC, De Kimpe C, Desjardins RL 2007: Agricultural production, greenhouse gas emissions and mitigation potential. *Agric. Forest Meteorol.,* **142**, 255–269.

Verhoeven JTA, Setter TL 2009: Agricultural use of wetlands: opportunities and limitations. *Ann. Bot*., 1-9. doi:10.1093/aob/mcp172.

Vizier JF 1990: Study of the dynamics of waterlogged environments: physcio-chemical approach. *Cah. ORSTOM, ser. Pedol*., **25**, 431–442.

Wahyunto, Dariah A, Agus F 2010: Distribution, properties, and carbon stock of Indonesian peatland. Proc.of Int. Workshop on Evaluation and Sustainable Management of Soil Carbon Sequestration in Asian Countries. pp 187-204. Bogor, Indonesia Sept. 28-29, 2010.

Wang B, Xu Y, Wang Z, Li Z, Ding Y, Guo Y 1999: Methane production potentials of twenty-eight rice soils in China. *Biol. Fertil. Soils*, **29**, 74–80.

Wang W, Sardans J, Laid DYF, Wang C, Zeng C, Tong C, Liang Y. Penuelas J 2015: Effects of steel slag application on greenhouse gas emissions and cropyield over multiple growing seasons in a subtropical paddy field in China. *Field Crops Research*, **171**, 146–156.

Wang ZP, DeLaune RD, Masscheleyn PH, Patrick Jr WH 1993: Soil redox and pH effects on methane production in a flooded rice soil. *Soil Sci. Soc. Am. J.*, **57**, 382–385.

Wardle DA, Nilsson MC, Zackrisson O 2008: Response to comment on "Fire-derived charcoal causes loss of forest humus". *Science*, **321**, 629.

Warren MW, Kauffman JB, Murdiyarso D, Anshari G, Hergoualc'h K, Kurnianto S, Purbopuspito J, Gusmayanti E, Afifudin M, Rahajoe J, Alhamd L, Limin S, Iswandi A 2012: A cost-efficient method to assess carbon stocks in tropical peat soil. *Biogeosciences*, **9**, 4477–4485

Wassmann R, Aulakh MS 2000: The role of rice plants in regulating the mechanism of methane emission. *Biol. Fertil. Soils*, **31**, 20–29.

Wassmann R, Lantin RS, Neue HU, Buendia LV, Corton TM, Lu Y, 2000: Characterization of methane emissions from rice fields in Asia. III. Mitigation options and future research needs. *Nutr. Cycling Agroecosyst*., **58**, 23–36.

Watanabe A, Purwanto BH, Ando H, Kakuda K, Jong F 2009: Methane and CO_2 fluxes from an Indonesian peatland used for sago palm (*Metroxylon sagu* Rottb.) cultivation: Effects of fertilizer and groundwater level management. *Agric. Ecosyst. Environ*., **134**, 14–18.

Watanabe T, Hosen Y, C, Agbisit R, Llorca L, Fujita D, Asakawa S, Kimura M 2010: Changes in community structure and transcriptional activity of methanogenic archaea in a paddy field soil brought about by a water-saving practice – Estimation by PCR-DGGE and qPCR of 16S rDNA and 16S rRNA. 19th World Congress of Soil Science, Soil Solutions for a Changing World, 1 – 6 August 2010, Brisbane, Australia.

Watson RT, Zinyowera MC, Moss RH, Dokken DJ 1996: Climate Change 1995, impacts, adaptations and mitigation of climate change: Scientific-technical analyses, Intergovernmental Panel on Climate Change. Cambridge University Press, USA. 879p.

West TO, McBride AC 2005: The contribution of agricultural lime to carbon dioxide emissions in the United States: dissolution, transport, and net emissions. *Agriculture Ecosyst. Environ*., **108**, 145–154.

Williams CJ, Yavitt JB 2010: Temperate wetland methanogenesis: the importance of vegetation type and root ethanol production. *Soil Sci. Soc. Am. J.*, **74**, 317–325.

Williarns RT, Crawford RL 1984: Methane production in Minnesota peatlands. *Appl. Environ. Microbiol*., **47**,1266–1271.

Wilson TW, Webster CP, Goulding KWT, Powlson DS 1995: Methane oxidation in temperate soils, Effects of land use and the chemical form of nitrogen fertilizer. *Chemosphere*, **30**, 539-546

Włodarczyk T, Stepniewski W, Brzezi´nska M 2002: Dehydrogenase activity, redox potential, and emissions of carbon dioxide and nitrous oxide from Cambisols under flooding conditions. *Biol. Fertil. Soils*, **36**, 200–206.

Wosten H, Hooijer A, Siderius C, Rais D, Idris A, Rieley J 2006: Tropical peatland water management modelling of the Air Hitam Laut catchment in Indonesia. *Int J River Basin Manage*., **4**, 233–244.

Wösten JHM, Clymans E, Page SE, Rieley JO, Limin LH 2008: Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. Catena, **73**, 212–224.

Xing GX 1998: N_2O emission from cropland in China. *Nutr. Cycl. Agroecosyst.*, **52**, 249–254, 1998.

Xiong ZQ, xing GX, zhu ZL 2007: Nitrous oxide and methane emissions as affected by water, soil and nitrogen. *Pedosphere*, **17**, 146-155.

Yagi K, Minami K 1990: Effect of organic matter application on methane emission from some Japanese paddy fields. *Soil Sci. Plant Nutr.,* **36**, 599–610

Yagi K, Tsuruta H, Kanda K, Minami K 1996: Effect of water management on methane emission from a Japanese rice paddy field: automated methane monitoring. *Global Biogeochem. Cycles,* **10**, 255-267

Yagi K, Tsuruta H, Minami K 1997: Possible options for mitigating methane emission from rice cultivation. *Nutr. Cycling Agroecosyst.*, **49**, 213–220.

Yan X, Akiyama H, Yagi K 2009: Global estimations of the inventory and mitigation potential of methane emissions from rice cultivation conducted using the 2006. Intergovernmental Panel on Climate Change Guidelines. *Global Biogeochem. Cycles.*, **23**, GB2002.

Yan X, Shi S, Du L, Xing G 2000: Pathways of N_2O emission from rice paddy soil. *Soil Biol. Biochem.,* **32**, 437-440

Yang C, Yang L, Ouyang Z 2005: Organic carbon and fraction in paddy soil as affected by different nutrient and water regimes. *Geoderma,* **124**, 133–142.

Yang C, Yang L, Yang Y, Ouyang Z 2004: Rice root growth and nutrient uptake as influenced by organic manure in continuously and alternately flooded paddy soils. *Agric. Water Manage*., **70,** 67–81

Yang J, Liu K, Wang Z, Du Y, Zhang J. 2007. Water-saving and high-yielding irrigation for lowland rice by controlling limiting values of soil water potential. *J. Integrative Plant Biology*, **49**, 1445–1454.

Yang J, Zhang J 2010: Crop management techniques to enhance harvest index in rice. J. *Experimental Botany*, 1-13. doi:10.1093/jxb/erq112

Yang JC, Huang D, Duan H, Tan G, Zhang J 2009: Alternate wetting and moderate drying increase grain yield and reduces cadmium accumulation in rice grains. *J. Sci. Food Agric.*, **89**, 1728–1736.

Yang SS, Chang EH 1998: Effect of environmental conditions on methane production and emission from paddy soil. *Agr, Ecosyst. Environ*., **69**, 69–80

Yao FX, Huang JL, Cui KH, Nie LX, Xiang J, Liu XJ, Wu W, Chen M, Peng S 2012: Agronomic performance of high-yielding rice variety grown under alternate wetting and drying irrigation. *Field Crops Res*., **126**, 16–22.

Yao H, Conrad R, Wassmann R, Neue HU 1999: Effect of soil characteristics on sequential reduction and methane production in sixteen rice paddy soils from China, the Philippines and Italy. *Biogeochemistry,* **47**, 269–295.

Yoshida S 1981: Fundamental of rice crop science. International Rice Research Institute (IRRI). Los Banos, Laguna, Philiphines. 269p.

Yu KW, Wang ZP,Vermoesen A, Patrick Jr WH, Van Cleemput O 2001: Nitrous oxide and methane emissions from different soil suspensions: effect of soil redox status. *Biol. Fertil. Soils*, **34**, 25–30.

Zhang A, Bian R, Pan G, Cui L, Hussain Q, Li L, Zheng J, Zhang X, Han X, Yu X 2012: Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a chinese rice paddy: A field study of 2 consecutive rice growing cycles. *Field Crops Res.*, **127**, 153–160.

Zhang A, Cui L, Pan G, Li L, Hussain Q, Zhang X, Zheng J, Crowley D 2010: Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. *Agric. Ecosyst. Environ.*, **139**, 469–475.

Zhang D, Pan G, Wu G, Kibue GW, Li L, Zhang X, Zheng J, Zheng J, Cheng K, Joseph S, Liu X 2015: Biochar helps enhance maize productivity and reduce greenhouse gas emissions under balanced fertilization in a rainfed low fertility inceptisol. doi.org/10.1016/j.chemosphere.2015.04.088

Zhang G, Zhang X, Ma J, Xu H, Cai Z 2011: Effect of drainage in the fallow season on reduction of CH_4 production and emission from permanently flooded rice fields. *Nutr Cycl. Agroecosyst.*, **89**, 81–91

Zhang H, Xue Y, Wang Z, Yang J, Zhang J 2009: Alternate wetting and moderate soil drying improves root and shoot growth in rice. *Crop Sci.*, **49**, 2246–2260.

Zhang H, Zhang S, Zhang J, Yang J, Wang Z 2008: Postanthesis moderate wetting drying improves both quality and quantity of rice yield. *Agron. J.*, **100**, 726–734

Zou JW, Huang Y, Jiang JY, Zheng XH, Sass RL 2005a: A 3-year field measurement of methane and nitrous oxide emissions from rice paddies in China: effects of water regime, crop residue, and fertilizer application. *Global Biogeochem. Cycles.*, **19**, 1–9

Zou JW, Huang Y, Lu YY, Zheng XH, Wang YS 2005b: Direct emission factor for N_2O from rice-winter wheat rotation systems in southeast China. *Atmos. Environ.*, **39**, 4755–4765.

ABBREVIATION

ABA	: abscisic acid	ECD	: electron capture detector
AEZs	: agro-ecological zones	Eh	: redox potential
Al_2O_3	: aluminum oxide	F	: fertilization
ANOVA	: analysis of variance	FDF	: flooded-drained-flooded
AWD	: alternate wetting and drying	Fe, Fe(II), Fe(III)	: iron
B	: boron	Fe_2O_3	: ferric oxide
BD	: bulk density	FID	: flame ionization detector
C	: carbon	GC	: gas chromatography
Ca	: calcium	Gg	: Giga-gram
CF	: continuous flooded	GHG	: greenhouse gas
$CH_3CO_2^-$	: acetate	GHGI	: greenhouse gas intensity
CH_4	: methane	GWP	: global warming potentials
Cl	: chlorine	H^+, H_2	: hydrogen
CO_2 eq	: carbon dioxide equivalent	H_2O	: water
CO_2	: carbon dioxide	H_2S	: hydrogen sulphide
Cu	: copper	I	: irrigation
D	: drainage	IAEA	: International Atomic Energy Agency
DAA	: days after amelioration		
DAT	: days after transplanting	IAERI	: Indonesian Agricultural Research Institute
DOI	: days of incubation		
DS	: dry season	ICALRRD	: Indonesian Center for Agricultural Land
EC	: electrical conductivity		

Resources Research and Development

IPCC : Intergovernmental Panel on Climate Change

IRRI : International Rice Research Institute

K : potassium

K_2HPO_4 : potassium hydrogen phosphate

K_2O : potassium oxide

KCl : potassium chloride

kPa : kilopascal

LULUCF : land use and land use change and forestry

Mg : magnesium

Mn, MnO : manganese

MnO_4 : permanganate

Mo : molybdenum

MRP : Mega Rice Project

mV : millivolt

N, N_2 : nitrogen

N_2O : nitrous oxide

NGHGI : National Greenhouse Gases Inventory of Indonesia

NH_2OH : hydroxylamine

NH_4^+ : ammonium

NO_2^- : nitrite

NO_3^- : nitrate

O_2 : oxygen,

P : phosphorous

P_2O_5 : super phosphate

Pg : Peta-gram

PVC : polyvinyl chloride

RS : rainy season

S : sulphur

SAS : statistical analysis system

S-AWD : site specific AWD

Si : silicon

SiO_2 : silicon dioxide

SNC : second national communication

SO_4^{2-} : sulphate

SOM : soil organic matter

TCD : thermal conductivity detector

Tg : Tera gram

Tukey's HSD test: Tukey's honest significant difference test

US EPA : United State Environmental Protection Agency

USDA : United State Department of Agriculture

WPI : water productivity index

Zn : zinc