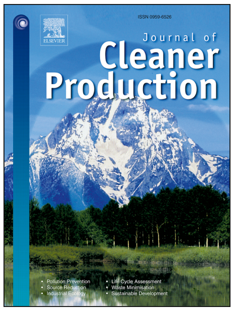
https://doi.org/10.1016/j.jclepro.2019.01.328

Alam, Md.K., Bell, R. and Biswas, W.K. (2019) ***Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains*. Journal of Cleaner Production**



Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains

Md. Khairul Alam, Richard W. Bell, Wahidul K. Biswas

PII: S0959-6526(19)30361-0
DOI: https://doi.org/10.1016/j.jclepro.2019.01.328
Reference: JCLP 15715

To appear in: *Journal of Cleaner Production*

Received Date: 17 September 2018
Revised Date: 28 January 2019
Accepted Date: 30 January 2019

Please cite this article as: Md. Khairul Alam, Richard W. Bell, Wahidul K. Biswas, Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains, *Journal of Cleaner Production* (2019), doi: 10.1016/j.jclepro.2019.01.328

Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains

Md. Khairul Alam[1,3], Richard W. Bell[1], Wahidul K. Biswas[2]

[1] Land Management Group, School of Veterinary and Life Sciences, Murdoch University, Murdoch, Western Australia 6150, Australia

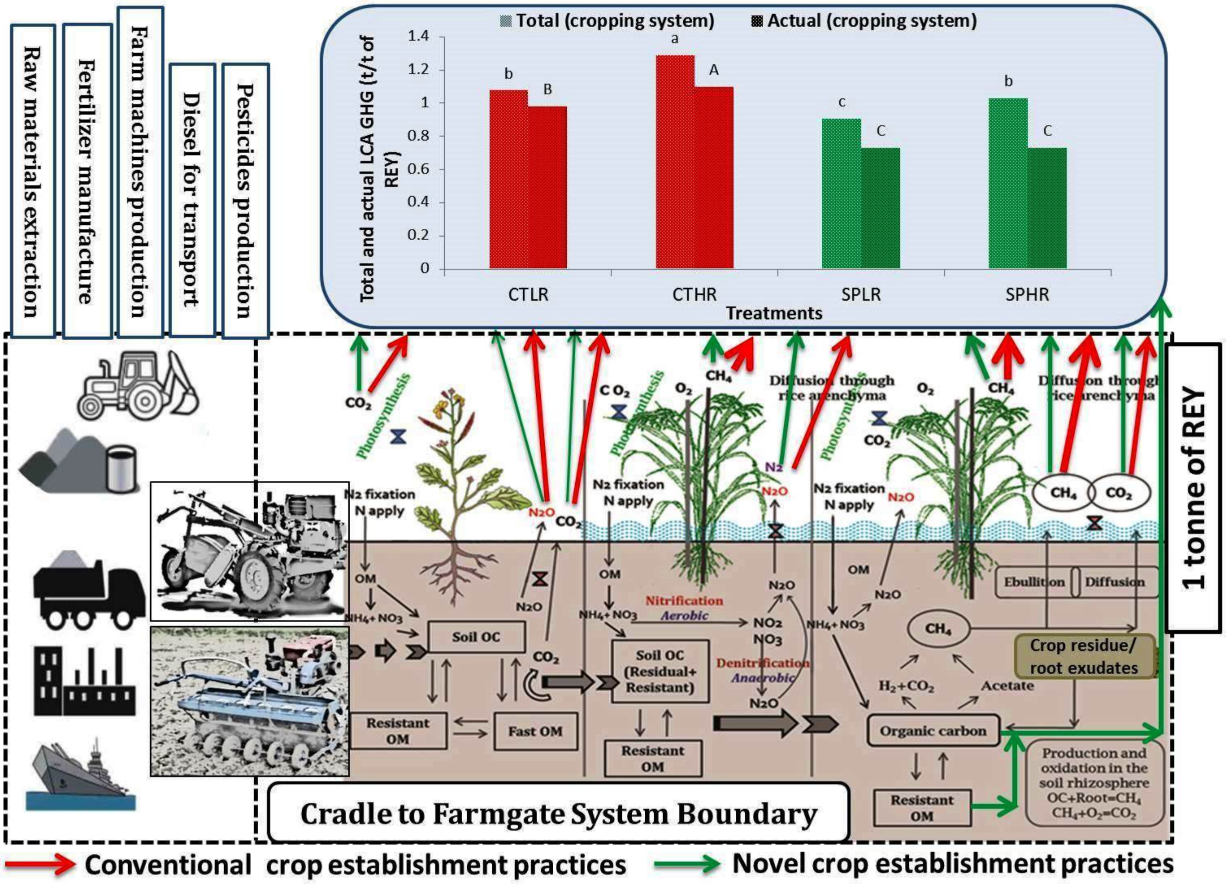
[2] Sustainable Engineering Group, School of Civil and Mechanical Engineering, Curtin University, Bentley, Western Australia 6845, Australia

[3] Soil Science Division, Bangladesh Agricultural Research Institute, Gazipur 1701, Bangladesh

Number of Tables: 4

Number of Figures: 7

Correspondence: Md. Khairul Alam; khairul.krishi@gmail.com; +8801761383121 and +61470318320



CT-Conventional tillage; SP-Strip planting; LR-Low residue retention; HR-High residue retention

Decreasing the carbon footprint of an intensive rice-based cropping system using conservation agriculture on the Eastern Gangetic Plains

Abstract

Emerging conservation agriculture (CA) technologies are being applied in rice-upland cropping systems and their potential to mitigate greenhouse gas emissions of the whole rice-based cropping systems could be significant in South Asia especially if they increase soil organic carbon (SOC) stocks. A streamlined life cycle assessment was conducted in the Eastern Gangetic Plains (Bangladesh) to determine greenhouse gas emissions from successive crops of monsoon rice (*Oryza sativa*), mustard (*Brassica juncea*) and irrigated rice under CA practices in contrast with the conventional crop establishment practice while accounting for changes in SOC. The life cycle greenhouse gas tonne^{-1} rice equivalent yield was assessed for four cropping practices: a) traditional crop establishment practices with farmers' practice of minimal residue return, or b) CT with return of increased residues; c) strip planting (for mustard)/ transplanting on non-puddled soils (for rice) with farmers' practice of minimal residue return or; d) strip planting/non-puddled transplanting with increased residue return. The global warming potential values for the 100-year timescale were used to calculate CO_2eq emissions within the system boundary. The net life cycle greenhouse gas emissions after allowing for changes in SOC sequestration varied from 0.73 to 1.12 tonne CO_2eq tonne^{-1} rice equivalent yield. In the annual cropping system, methane (CH_4) released from on-farm stage of the life cycle assessment, particularly from the rice crops, represented the dominant contributor to life cycle greenhouse gas emissions. The greenhouse gas emitted by machinery usage during the on-farm stage (irrigated rice), CO_2 emission from soil respiration (monsoon rice), and greenhouse gas related to manufacture of inputs (mustard) were secondary sources of emission, in that order of priority. The non-puddled transplanting of soil with low and increased residue retention were the most effective greenhouse gas mitigation options when sequestered SOC was taken into account (they avoided 35 % of the net life cycle footprints compared with current farmers' practice) in footprints of component crops of the rice-upland cropping system. The CA approaches being developed for the Eastern Gangetic Plains involving strip planting or non-puddled transplanting of rice have potential to mitigate global warming potential of intensive rice-based triple cropping systems but the life cycle assessment approach needs to be applied to a more diverse range of rice-based cropping systems.

Key words: Crop establishment practices; labour requirement; life cycle GHG; non-puddled transplanting; puddled transplanting; rice-upland triple cropping system; strip planting

Abbreviations:

ACIAR–Australian Centre for International Agricultural Research

ADB–Asian Development Bank

BBS–Bangladesh Bureau of Statistics

CA–Conservation agriculture

C–Carbon

CI– Carbon intensity

CH_4–Methane

CO_2–Carbon dioxide

CO_2eq–Carbon dioxide equivalent

CS LCA GHG – Cropping system life cycle assessment greenhouse gas

CT–Conventional puddling

DECC–Department of Energy and Climate Change

DEFRA–Department for Environment, Food and Rural Affairs

DSR–Direct-seeding of rice

Eh–Redox potential

EGP–Eastern Gangetic plains

GHG–Greenhouse gas

GoB–Government of Bangladesh

GWP–Global Warming Potential

ha–Hectare

HR–High residue retention

IEA–International Energy Agency

IFA–International Fertilizers Association

IPCC–Inter–Governmental Panel on Climate Change

ISO–International Organization of Standardization

LCA–Life Cycle Assessment

LCI–Life Cycle Inventory

LSD–Least significant difference

LR–Low residue retention

MOEF–Ministry of Environment and Forest, Peoples Republic of Bangladesh

MoP–Muriate of potash

N–Nitrogen

N_2O–Nitrous Oxide

SOC–Soil organic carbon

SPSS–Statistical Package for the Social Sciences

t–Tonne

TOC–Total organic carbon

UN-FCCC–United Nations Framework Convention on Climate Change

NP–Non-puddled transplanting of rice (A rice crop establishment practice that avoids wet-tillage operations. It limits soil disturbance to narrow strips and is compliant with the FAO conservation agriculture definition).

NT–No-tillage

US$–United States Dollar

USA–United States of America

1. Introduction

In order to inform cleaner production technologies, energy balance (Iqbal, 2007; Abbas, 2011; Heidari and Omid, 2011; Lal et al., 2015) and greenhouse gas (GHG) fluxes have been reported for various cropping and soil management practices in single crops in numerous studies (Harada et al., 2007; Wang et al., 2010; Gan et al., 2012a; Gan et al., 2012b; Thanawong et al., 2014; Vetter et al., 2017). Hokazono and Hayashi (2015) found that diversifying crop rotations with non-rice/leguminous crops can mitigate life cycle GHGs. Similarly pulse crops have low carbon (C) and water footprints relative to most field crops, with GHG emissions of 0.27 kg CO_2eq kg^{-1} and irrigation water use of 0.19 m^3 kg^{-1} (Gustafson and Yildiz, 2017). Liu et al. (2016) estimated C intensity (CI) of rice in corn-rice system over three consecutive rice-growing cycles from year 2011 to 2013; the corn-straw amendment at increased rate had a much higher CI of the following rice crop (2.49 kg CO_2eq kg^{-1} grain) than that of no-amendment (0.88 kg CO_2eq kg^{-1} grain), resulting from large soil CH_4 emissions.

The rice-based intensive cropping systems which are critically important to food security in the EGP warrant further study in terms of GWP mitigation, as there is limited information on the C footprint of complete crop rotations (Alam et al., 2016). The hotspots contributing the largest emissions to the life cycle GHGs are known to vary among crops due to differences in irrigation, fertiliser rates, climate, fuel use for transportation and farm machinery use, varied yield per unit area and harvesting techniques (AIJ, 2003; Lal, 2004; Brentrup, 2009). Since farmers across South Asia grow diverse sequences of crops, it is important to determine the GWP of complete annual crop rotations rather for single crops.

Rice–based triple cropping systems have complex effects on GHG emissions due to variation during the year in temperature and water regimes, varied duration of crop growth, as well as differences in crop outputs (and yields), energy/feedstock use efficiencies, nutrient (fertilizer) inputs, residue/carbon returns and other inputs influencing management activities and production. Tillage practices, residue management, growing crops in rotation, chemical use for pest and nutrient management control the global emissions of GHGs and eventually the SOC sequestration also (Duxbury et al., 1993). Soil disturbance by tillage may decrease topsoil OC, but inversion tillage may bury the C-enriched soils and organic materials that can increase it in the deeper layers (Vanden Bygaart and Angers, 2006). By contrast, minimum soil disturbance and residue retention might enhance accumulation of SOC by increasing biomass yields and simultaneously by slowing down SOC loss (Lal et al., 1999; Alam et al., 2018). So et al. (2001) showed that deployment of conservation tillage, in many situations, can reduce CO_2 emissions from the soil by 4.3 Mt yr^{-1}, compared with emissions from soils under conventional tillage. Soil disturbance has mixed effects on N_2O emissions (minimum tillage accounted for higher N_2O emission (Ussiri et al., 2009), insignificant emission (Jantalia et al., 2008) or lower N_2O emission (Steinbach and Alvarez, 2006) than emissions in soils under zero tillage). In addition, incorporation of residues into the field may increase sequestered SOC but enhance CH_4 fluxes (Sass, 2002). Lower CH_4 emission by mid-season drying of rice soils due to drainage may be accompanied by increased emission of N_2O (Towprayoon et al., 2005). Harada et al. (2007) reported the net life cycle GHG up to milling (brown rice) for puddling, no-tilling and non-puddling were 0.94, 0.44 and 0.76 t CO_2eq t^{-1}

brown rice. The non-puddling practice adopted in the study of Harada et al. (2007) involved conventional tillage and planting without puddling. By contrast, non-puddled transplanting of rice following minimal disturbance of soil (strip tillage) in a rice-based triple cropping system (where other upland crops are growing following strip planting) has performed well in both biogenic GHG and life cycle GHG reduction under flooded, irrigated conditions (Alam et al., 2016). However, the CA practices applied to all crops in the rice-based cropping system have not been studied for estimating its contributions to life cycle GHG.

The variations in C addition to soils, level of disturbance of soils, and decomposition of residue define whether there is a net storage of SOC or loss from soil. The cultivated crops supply variable C inputs from the turnover of roots, return of crop residue and live roots, and root exudates (Ghimire et al., 2017). Soil C inputs from aboveground biomass, belowground root biomass, root exudates and rhizodeposits, microbial biomass and organic amendments help increase SOC sequestration, while growing crops in rotation under minimum soil disturbance and return of crop residues at increased rates enhance the processes of SOC gain (Freibauer et al., 2004). In addition, recent development of non–puddled transplanting of rice and strip planting for upland crops in rice-based cropping systems, has been reported to increase SOC (0-10 cm) while reducing fuel use and life cycle GHGs per unit of irrigated rice yield (Alam et al., 2016, 2018). But these studies only considered a single rice crop. Inclusion of SOC sequestration data is also very important to estimate the net C footprint of crop production alone or in a system (Alam et al., 2018; Alam et al., 2019). Lal (2004) stated that SOC sequestration counterbalances fossil fuel emission of GHGs. The SOC sequestration can offset the high CH_4 emission in part, while accounting for soil sequestered C in LCA study of a long-term cropping system is critically important for finding actual/net life cycle GHGs for any crop production practices (Goglio et al., 2015; Petersen et al., 2013). Soil carbon sequestration accounting is necessary for estimating the net contribution of novel crop or soil management practices that alter SOC over time otherwise there will be an overestimation of GHG emissions (Marble et al., 2011).

Given the complexity of factors affecting growth of each crop of the rice-based triple cropping system in a year, an accounting of net life cycle GHG fluxes together with C sequestration in soil, is needed to evaluate strategies of GWP mitigation for rice-dominant cropping which is a major contributor to the C footprint of global agriculture (Robertson and Grace, 2004). The current research was conducted to:

1. develop a complete LCA of a mustard–irrigated rice-monsoon rice cropping system practiced in the EGP by taking SOC into account
2. To highlight the hotspots in the cropping system life cycle GHG emissions, and
3. To determine the relative contributions of each crop to emissions of major GHGs within the cradle to farmgate boundary of the cropping system LCA.

2. Materials and methods

2.1 Details of experimental location and design

Details of the study site and experimental design are available from Alam et al. (2016), Alam et al. (2018) and Alam et al. (2019). Relevant details of the experimental site are given in Table 1.

The field study covered three crops namely, mustard in the dry winter (*rabi* season from 18 November to the 18 February), irrigated pre-monsoon rice (kharif-I stretching from 7 March to the end of 20 June) and rainfed monsoon rice (kharif-II stretching from July 19 to mid-October 15). The experiments adopted conventionally puddled (CT) and non-puddled (NP) rice establishment practices for rice (see Haque et al., 2016 for further details of non-puddled transplanting), both with increased residue return (HR) and low residue return (LR) as treatments (Table 2). Having been established in 2010, the long-term experiment continued for five years with four replicates of each practice in a split plot design (Islam, 2017). The low residue retention practices resembled practices followed in the region by farmers and increased residue (HR) practice retained 50 % of standing rice residues by height. The HR practice for all the previous mustard, mung bean (*Vigna mungo* L.) and lentil (*Lens culinaris* L.) crops involved return of all residues of the crops to the respective sub plots as mulch. By contrast, LR for mustard, lentil and mung bean involved complete removal of aboveground biomass. Lentil, mung bean and monsoon rice were grown on the field in a sequence for the first three years, whereas mustard, irrigated rice and monsoon rice were grown in a sequence in the following three years on the same field. Fertilisers for crop nutrition and pesticides for crop protection were characteristic of the practice followed in the locality (see Table 3). The GHG emissions (CO_2, CH_4 and N_2O) from soil were measured with static chambers similar to the studies of Naser (2005) and Alam et al. (2016). The gases were sampled from each subplot with a static closed chamber during rice seasons. The samplings were repeated every 7 days throughout the growing period of each crop (Alam et al., 2016). The measurement frequency for GHGs was increased to 2 or 3 days during application of split doses of N. The gas samplings were done at 1, 7, 15, 22, 30, 45, 60, 75, 90 and 100 days for mustard. The gas samples were analysed using gas chromatography for CO_2, CH_4 and N_2O with a CO_2 detector, hydrogen flame ionized detector and combined gas analyzer, respectively (Naser, 2005).

Table 1. Summary of the characteristics of the life cycle GHG study site

Characteristics of study site	Details
Location	Northwest Bangladesh at Alipur village, Durgapur upazilla, Rajshahi division
Texture class	Silty loam
Soil type	Calcareous Brown Flood Plain
Subgroup (US Soil Taxonomy)	*Aeric Eutrochrept*
Parent material	Ganges river alluvium
Location (Latitude and longitude)	24° North latitude, 88° East longitude.
Landform	Narrow terraced strips on the gently undulating slopes of the flood plain.
Altitude	8 m above sea level
Rainfall	1047 to 1693 mm; lower than other parts of Bangladesh; concentrated on monsoon season (June to September)
Dominant minerals	Mica–vermiculite–smectite (interstratified) and kaolinite–smectite (interstratified), Mica, Kaolinite (Moslehuddin et al., 2009)
Drainage	Moderate

mm=9illimeter; m=metre; USDA= United States Department of Agriculture

306 **Table 2.** The field treatments at the Alipur and Digram sites.

Crops	Soil management	Residue management
Mustard	• Conventional tillage (CT) • Strip tillage (SP)	• Low residue retention (LR) • Increased residue retention (HR)
Irrigated rice	• Conventional puddling (CT) • Non-puddling (NP) followed by SP	• LR • HR
Monsoon rice	• CT • NP	• LR • HR

307

308 **Table 3.** Life Cycle Inventory of farm activities, inputs and outputs for the production of one 309 tonne of rice or rice equivalent yield (REY) on the Eastern Gangetic Plain

Treatments	**CTLR**[a]	**CTHR**[b]	**NPLR**[c]	**NPHR**[d]
a) Seeds and chemicals (kg tonne^{-1} of rice production or REY)				
1. Seeds	7.43	7.00	7.19	6.63
2. Nitrogen	22.41	20.80	21.72	19.74
3. Phosphorus	10.11	9.49	9.77	9.00
4. Potassium	15.39	14.44	14.87	13.73
5. Sulfur	4.94	4.64	4.78	4.40
6. Zinc	0.83	0.78	0.80	0.74
7. Boron	0.39	0.34	0.35	0.32
8. Fungicides	0.22	0.21	0.21	0.19
9. Herbicides	0.56	0.52	0.54	0.50
10. Insecticides	0.38	0.36	0.37	0.34

b) Transport (km for road + t–nm for sea)[1]				
1. Urea	65.3	61.3	63.1	58.2
2. Triple superphosphate	86.33+565.3	81.2+531	83.5+546.8	77+503.5
3. Muriate of potash	86.33+394.8	81.2+370.3	83.5+382	77+352.1
4. Gypsum	86.33+394.8	81.2+370.3	83.5+382	77+352.1
5. Zinc	86.33+394.8	81.2+370.3	83.5+382	77+352.1
6. Boric acid	86.33+275.3	81.2+259.2	83.5+266.6	77+245.8
7. Insecticides	68.9	64.7	66.6	61.4
8. Fungicides	63.6	60.3	63.6	60.3
9. Herbicides	86.3+179.7	81.2+168.5	83.5+173.8	77+160.2
c) Farm machinery (US$ tonne^{-1} of REY production)				
1. Power tiller/VMP	0.104	0.103	0.048	0.048
2. Harvester	0.047	0.039	0.045	0.038
3. Irrigation pump	1.92	1.81	1.87	1.72
d) Farm machinery transport (km for road + t–nm for sea)				
1. Harvester	86.3+275.3	81.2+259.2	83.5+266.7	77+245.7
2. Power tiller	86.3+275.3	81.2+259.3	0	0
3. VMP	0	0	83.5+266.6	77+245.8
4. Irrigation pump	57.6+183.6	54.1+172.9	55.7+177.7	51.3+163.9
d) On–farm (litre tonne^{-1} of REY production)				
1. Rotary tiller/versatile Multicrop Planter (VMP)	2.48	2.39	0.99	0.95
2. Harvester	28.68	29.35	27.52	28.52
3. Irrigation pump	23.22	23.37	20.98	20.70

e) Soil emission (kg per tonne of REY production)				
CO_2	188.7	248.0	160.0	193.7
CH_4	23.6	30.1	19.3	23.3
N_2O	0.16	0.15	0.12	0.11
Soil C-sequestration (0-30 cm)	96.38	164.4	179.8	300.9
REY (t ha^{-1})	18.16	19.33	18.78	20.38

[1] t–nm=tonne–nautical mile; [a]puddled transplanting with low residue retention (CTLR); [b]puddled transplanting with high residue retention (CTHR); [c]non-puddled transplanting with low residue retention (NPLR) and [d]non-puddled transplanting with high residue retention (NPHR).

2.2 Soil sampling method and soil C sequestration estimation

The carbon sequestered in soils due to the continual application of the above treatments was included in the carbon accounting. Soils at 0-30 cm depth from each treatment were collected, assessed for bulk density and analysed for SOC content. In this study, C sequestration estimation at 0-30 cm depth only used data from crop 15 to crop 18 to represent recent trends because the rate of SOC accumulation during the initial years of CA establishment and after three years may not be the same. The organic carbon content measured by wet oxidation method (Jackson, 1973) was used to calculated total organic carbon (TOC) content (Ellert and Bettany, 1995; Alam et al., 2018).

The increments in TOC over three years were then divided by the number of crops to approximate the C accumulated over a single crop growing season as well as over the cropping system.

2.3 GHGs measurement and gas flux calculations

A detailed description of gas sample collection for measuring CO_2, CH_4 and N_2O emissions under irrigated rice was reported in Alam et al. (2016). The following variations were used for monsoon rice (Alam et al., 2018). For measuring CH_4 and N_2O, triplicate 5 mm thick transparent chambers constructed with acrylic material sheets (plexiglas) were installed in each plot with the dimension of 60 cm length × 30 cm width × 100 cm height. The measurement of soil CO_2 efflux as a product of heterotrophic respiration were done with the chamber of dimensions 30 cm length × 30 cm width × 60 cm height made with acrylic materials (3 mm thick sheets) (Hutchinson and Livingston, 1993).

For heterotrophic respiration measurement in soil under the mustard crop, circular chambers with the dimension 100 cm height × 20 cm diameter were established in each plot (triplicated). Vials containing 35 mL of 0.5 M NaOH were used to trap evolved CO_2. The replacement of vials was repeated at intervals of 2-3 days up to 15 days after placement of chambers. With the decreased rate of CO_2 evolution after 15 days, the frequency of replacement of vials was reduced to an interval of 5-7 days. In addition, the amount and concentration of NaOH were reduced to 30 mL of 0.25 M. The CO_2 entrapped with NaOH was measured according to Anderson (1982) (10% w/v $BaCl_2$). According to Alam et al. (2018), CO_2 from the control treatment was trapped and measured with 35 ml of 0.5 M NaOH solution in a chamber without soil and this value was subtracted from CO_2 released from soils of each practice (treatment). For mustard crops, N_2O emission induced by N fertilizer (organic or synthetic) was estimated following the IPCC (2006) recommendation, i.e. 0.01 t N_2O–N t^{-1} fertiliser-N or N in organic amendment (or any source), for dry cropland.

The calculation of gas flux over the crop growing season was done in line with Yagi et al. (1991). It was assumed that GHG emissions fluctuated linearly during the periods between gas sampling times. Consequently, the total GHG fluxes over the growing seasons were summed from the average gas emissions as done by Alam et al. (2016).

2.4 Life cycle GHG emissions of the irrigated rice-monsoon rice-mustard crop rotation

The LCA approach was used to estimate GWP with the inclusion of GHG emissions only (Alam et al., 2016). Also called a streamlined LCA analysis, the study took into account GHGs emanating from cradle–to–farm gate stages of each crop of the cropping system (ISO, 2006). According to ISO 14040–44, the four steps of the streamlined LCA approach, namely defining goal and scoping, preparing life cycle inventory (LCI), assessing impact and interpreting results, were considered for estimation of the GHG emissions of each crop. Like Finkbeiner et al (2011), this study can also be considered a limited focus LCA that considered one impact category only, i.e. global warming impact (GWI). The LCA method and principles can be applied to estimate GWI in terms of GHG emissions, as evidenced by other recent literature that estimated the GHG emissions of agricultural products (Biswas et al. 2008; Alam et al. 2016; Denham et al. 2016).

Net life cycle GHGs were calculated by subtracting the CO_2–eq for SOC sequestered during the rice crop from the total carbon footprint of the product.

2.4.1 Goal setting and scope definition

The emissions of GHGs associated with the production of component crops for the rice-based cropping system were calculated based on the cropping practices as shown in Table 2.

The system boundary for each crop of the cropping system of the study was determined up to farm-gate (pre-farm and on-farm stages). The functional unit of the LCA is one tonne of rice or rice equivalent yield (REY) of the cropping system (Figure 1). When yield of one crop is converted into equivalent yield of rice crop, it is called rice equivalent yield. Rice equivalent yield of different crop sequence is calculated by multiplying the grain yield of crops with their respective price unit^{-1} of land (ha). The REY of mustard in the cropping system was calculated as follows (Anjeneyul et al., 1982):

$$\text{REY} = \text{Rice yield} + \frac{\text{Mustard crop yield} \times \text{Market price of mustard crop}}{\text{Market price of rice}}$$

A mass balance approach was also used to estimate the inputs and outputs per tonne production of REY within the system boundary (up to farmgate), which is known as a life cycle inventory. The GHGs associated with the pre-farm activities were estimated by multiplying the emission factors (EF) with the amount of inputs required for their production and transportation to the field of the current study, while GHGs emanated by on-farm activities are outputs associated with operating farm machinery and applying chemicals. The total GHG emission from the production of one tonne of REY was calculated by adding emissions from both the stages (pre- and on-farm) of each crop (Figure 1).

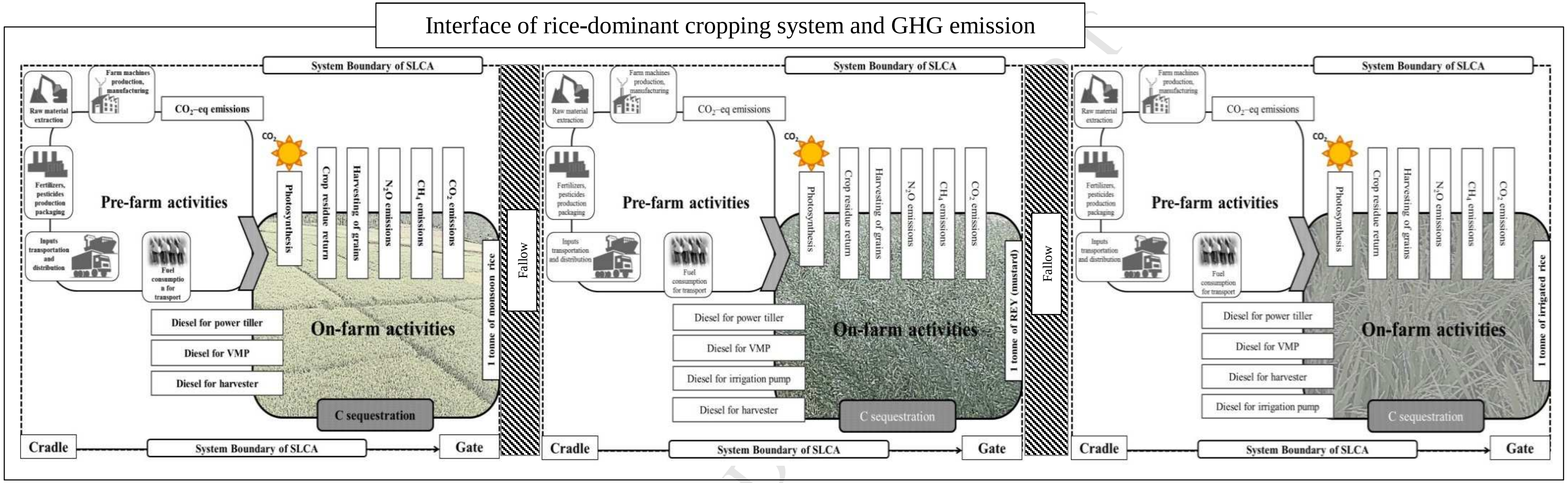


Figure 1. System boundaries of a rice–based triple cropping system and input–output relationship adopted in the streamlined life cycle analysis (SLCA). Here, VMP-Versatile Multi-crop Planter, CO_2-Carbon dioxide, CH_4-Methane, N_2O-Nitrous oxide.

2.4.2 Life cycle inventory

A LCI was formed by using the factors related to the production of each tonne of REY (e.g., chemicals for crop nutrition and crop protection, machinery) for estimating the GHGs for the manufacturing, transport and use of inputs and outputs. Soil emissions (CO_2, CH_4 and N_2O) are positive outputs and SOC-sequestration is a negative output (Table 3) of REY within the system boundary of this study.

2.4.2.1 Pre–farm emissions

Greenhouse gas emissions of activities related to input production and their delivery up to farm were estimated. Based on the LCA study conducted for irrigated rice production, indirect emissions from manufacturing of farm machinery were calculated by following the database of input/output of USA (Suh, 2004) as described by Alam et al. (2016). The EF of farm machinery production (0.15 kg CO_2eq US$$^{-1}$) was multiplied by the cost of machinery manufacture for each functional unit determined according to 1998 US$ value (WB, 2014).

The chemicals used for mustard and rice production following the establishment practices under study were recorded per tonne of rice or REY production. These EFs were sourced from Alam et al. (2016) as they represent the general condition in Northwest Bangladesh. The EFs of crop nutrients used from Alam et al. (2016) were for fertilizers (urea, triple superphosphate), crop protection insecticides (Malathion™, Sumithion™), fungicides (Amistar™ and Tilt™) and herbicides (Refit™ and glyphosate). For the insecticide, Wonder 5WG (Emamectin Benzoate), and fungicide, Rovral 50WP (Ipridione), the local EF was determined by multiplying EFs of local level production of energy with the embodied energy consumption (RMIT, 2007; DEFRA, 2008) of these chemicals. The GHG EFs of urea, superphosphate and pesticide production were sourced from the work of Alam et al. (2016) who considered the EF for electricity generation was 0.64 kg CO_2eq kWh^{-1} following UN–

FCCC (2017). The source countries of imported inputs were collected from Bangladesh Business News (2013) and BBS (2013), while the EFs of the inputs imported to Bangladesh (urea, triple superphosphate, muriate of potash (MoP), gypsum, zinc sulphate monohydrate and boric acid) were obtained from Alam et al. (2016) as their represent the overall situation of the study area.

The data of emissions regarding transportation of materials for rice and mustard production were sourced from available databases (INFRAS, 2010; WRI and WBCSD, 2013 and HBFEA, 2014). The modes employed for transportation include the transportation by sea (trans-oceanic bulk cargo carrier) and trucks (3–7 tonnes) for road transport. The emission of GHGs for input deliveries from factory to crop field are expressed in terms of tonne-kilometres (tkm) travelled by road and tonne-nautical miles (t-nm) travelled by sea. The distance between the paddy field and its source was multiplied by the weight of input to determine 'tkm' (Lal, 2004; Alam et al., 2016 and Zhang et al., 2017).

2.4.2.2 On–farm emissions

On-farm greenhouse gas emissions started with land preparation for establishment of each the three crops. The emissions further include soil emissions after application of chemicals for crop nutrition and protection and intercultural operations and finally fuel use for harvesting.

Farm machinery–In the case of the conventional system, a rotary tiller was used for land preparation for the establishment of rice crops following puddling of soil, and a strip planter was used to prepare strips for transplanting rice into non-puddled soil and for sowing mustard seeds (Haque et al., 2016). A harvester of 9 kW was used for harvesting rice. Fuel consumption in terms of litres per hectare by the farm machinery was measured during farming operations and was dependent on area of land, operating width of machinery (tiller and harvester) and the number of machinery passes across the land (Alam et al., 2016). The EFs of fuel combustion for the usage of light machinery (≤500kW) were collected from RMIT (2007), INFRAS (2010) and HBEFA (2014) and these values were used to calculate GHG emissions. The light machineries considered for this experiment are commonly used in this EGP region. The fuel use (litres ha^{-1}) was based on machinery usage in standard machinery terms of the region (for Versatile Multi-crop Planter 1.25, for rotary tiller 3.22 to 3.32 and for harvester 1.82 to 2.11 L t^{-1}).

Soil – The major GHGs (CO_2, CH_4 and N_2O) emitted directly from soil of the experimental site were measured as detailed in the GHGs measurement and gas flux calculations section above. The emissions of N_2O that occur indirectly via volatilization of ammonia and leaching of nitrate were excluded from the study owing partly to lack of data. In addition, for this soil, occurrence of a hard pan beneath the plough layer (Islam, 2017) restricts leaching loss of N to the deep soil layers (Patil and Das, 2013). Moreover, there was continuous standing water in the field for rice production (Alam et al., 2016 and Alam et al., 2019) which lowers the level of nitrate and therefore the risk of synthesis of N_2O via denitrification during rice (Dobbie and Smith, 2006).

2.4.3 Impact assessment

The input and output data in the inventory (Table 3) were multiplied by the corresponding EFs (Table 4) in order to estimate GHG emissions associated with the production of 1 tonne of rice or its equivalent. These GHG emissions were then converted to CO_2 equivalent GHG emissions or GWI using an IPCC method.

Table 4. Different inputs required for growing three crops in irrigated rice-monsoon rice-mustard cropping system, their emission factors and sources of data

Input		Emission factor	Comment/References
Fertilizer			
	Urea-N	5.5 kg CO_2/kg N	Alam et al., 2016; IFA, 2009; IEA, 2012
	TSP-P	0.34 kg CO_2/kg P	Alam et al. 2016; IFA, 2009; IEA, 2012
	MoP-K	0.58 kg CO_2/kg K	Alam et al. 2016; IFA, 2009; IEA, 2012
	Gypsum-S	0.3 kg CO_2/kg S	Wells, 2001; Saunders et al., 2006
Herbicides			
	Glyphosate	33.4 kg CO_2/kg a.i.	DEFRA, 2008; Bosch and Kuenen, 2009; Brander et al., 2011
	Refit 50EC	16.1 kg CO_2/kg a.i.	DEFRA, 2008; Bosch and Kuenen, 2009; Brander et al., 2011
Fungicides			
	Amistar 250EC (Propiconazole)	17.5 kg CO_2/kg a.i.	Lal, 2004; Brander et al., 2011
	Tilt 250EC (Propiconazole)	17.3 kg CO_2/kg a.i.	Lal, 2004; Brander et al., 2011
	Rovral 50WP (Ipridione)	16.9 kg CO_2/kg a.i.	RMIT, 2007; DEFRA, 2008
Insecticides			
	Malathion (Organophosphorus)	17.7 kg CO_2/kg a.i.	Alam et al., 2016; Brander et al., 2011
	Sumithion (Organophosphorus)	17.7 kg CO_2/kg a.i.	Alam et al., 2016; Brander et al., 2011
	Wonder 5WG (Emamectin Benzoate)	17.7 kg CO_2/kg a.i.	Alam et al., 2016; Brander et al., 2011
Vehicle	Light-duty diesel truck	2.85 kg CO_2/L	HBEFA, 2014; World Resource Institute and WBCSD, 2013).
	Trans-oceanic freighter	14.5 g CO_2/t-nm	UN–FCCC, 2017; Biswas et al., 2008; Barton et al., 2014
Electricity	Electricity Generation	0.64 kg CO_2eq kWh^{-1}	Brander et al., 2011; Suh 2004
Machinery	Farm machinery production	0.15 kg CO_2eq US^{-1}	Suh, 2004; Barton et al., 2014
Fuel	Fuel use (Diesel)	3.1 kg CO_2/L	HBFEA, 2014; Lal, 2004

A GWI value for the 100-year time horizon was used to estimate the CO_2 equivalent GHG emissions for the production of each functional unit (1 tonne of rice or REY) of each crop of the cropping system. The conversion factors used for converting CH_4 and N_2O to the baseline unit, CO_2, were 25 and 298 (IPCC, 2013). The total CO_2eq emission (kg CO_2eq ha^{-1}) was summed for all the studied cropping seasons covering the span of one year, excluding the fallow periods between crops. The net life cycle GHGs for production of each unit (one tonne) of REY were then calculated by subtracting the sequestered SOC over the year of study (three seasons) from the total life cycle GHGs.

2.5 Formulae adopted to determine the net C footprint of each crop of the rice-based triple cropping production system:

Net life cycle GHG = Total life cycle GHG− SOCSR

Net life cycle GHG (kg C-eq ha^{-1}) is the emissions of GHG (carbon equivalent) from producing crop in a unit of land (ha^{-1}), SOCSR denotes sequestered C from the same unit of land (ha^{-1}) at 0-30 cm depth by the production of crop (kg ha^{-1}).

2.6 Statistical analysis

The effects of soil disturbance for crop establishment and residue return on the CO_2eq emission from pre-farm, on-farm and total and net life cycle GHG emissions and sequestered SOC were statistically analysed with a two–factor split plot analysis of variance by using SPSS software v21 (SPSS Inc., Chicago, IL, USA). Least significant difference (LSD) value was adopted in comparing means at a 5 % significance level.

3. Results

3.1 GHG emission for rice-based triple cropping system

Crop establishment practices and residue retention levels changed life cycle GHG emissions of the rice-dominant cropping system (CS LCA GHG-the carbon footprint estimated for a cropping system per tonne of rice or REY) comprising irrigated rice, monsoon rice and an upland mustard crop in the rotation ($p<0.05$; Figure 2, 3 & 4). Irrespective of the crops and growing seasons, NP/SPHR emitted the lowest cropping system life cycle GHG t^{-1} of rice (or REY) production, followed by NP/SPLR and CTLR, respectively. The performance of NP/SPHR and NP/SPLR was more similar when sequestered C over five years of CA cropping was accounted for: these practices saved 34.5 % and 50.8 % and 34.4 % and 50.7 % cropping system life cycle GHG compared to CTLR and CTHR, respectively. In case of mustard, SPHR practice had 76 %, 69 % and 35 % lower net life cycle GHG emissions than life cycle GHG recorded under CTLR, CTHR and SPLR practices, respectively, whereas, for irrigated rice, the NPHR had 36.8 % and 54.6 % and 4.4 % lower net life cycle GHG than those recorded under CTLR, CTHR and NPLR practices, respectively (Figure 2. In case of monsoon rice, NPLR saved 31.4 %, 51.9 % and 14.2 % net life cycle GHG than those under CTLR, CTHR and NPHR practices, respectively (Figure 2)

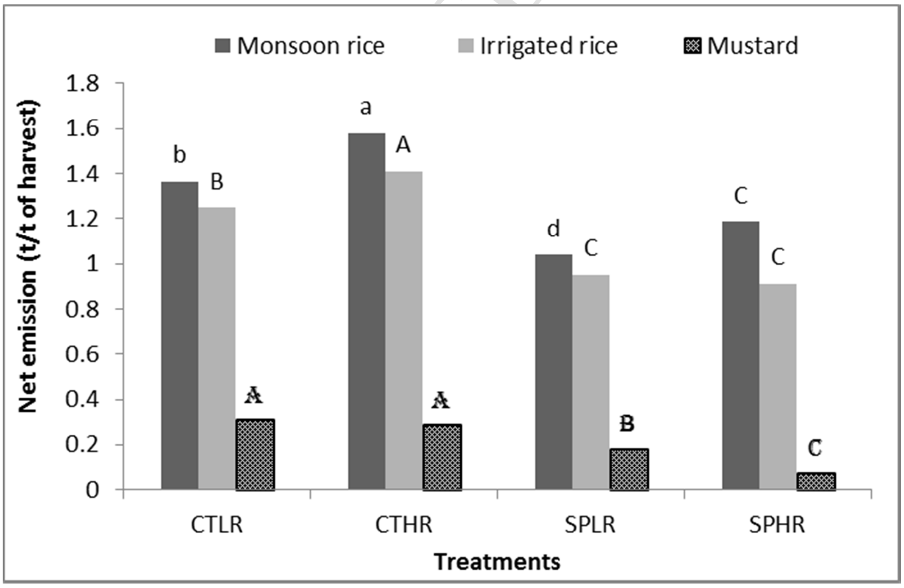


Figure 2. Net life cycle GHG emissions per season for the production of one tonne of component crops (irrigated rice, monsoon rice and mustard: in the case of mustard, rice equivalent yield was used) as influenced by crop establishment techniques and residue retention ($p<0.05$). Bars with the same letter above them for a specific crop are not significantly different at $p < 0.05$. Comparisons are made among emissions converted to CO_2–eq according to global warming potentials of CO_2, CH_4 and N_2O over 100-year time horizons. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

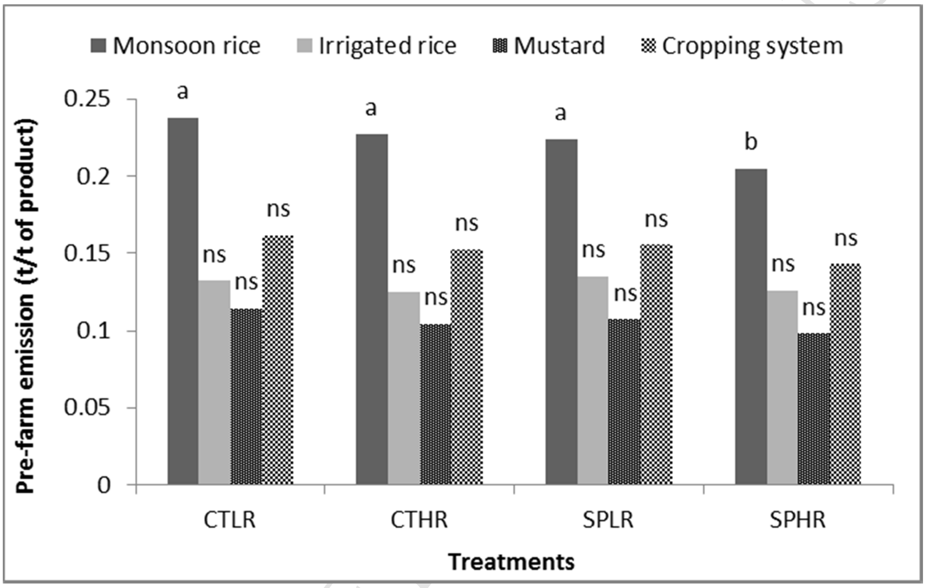


Figure 3. Pre-farm life cycle greenhouse gas emissions produced per season for one tonne of rice, mustard and cropping system production (the mustard and cropping system as rice equivalent yield) as influenced by crop establishment techniques and residue retention ($p<0.05$). For a particular species or the cropping system, bars with the same letter above them are not significantly different at $p<0.05$. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

The life cycle GHG emitted by the winter mustard crop (after conversion to REY) under any of the crop establishment practices was lower than life cycle GHGs emitted during wetland crops under the same practice. As for example, life cycle GHGs of mustard cultivation under SPLR were 71 % and 69 % lower than life cycle GHGs of monsoon rice and irrigated rice, respectively.

In irrigated rice and mustard grain production, the emissions associated with different crop establishment and residue retention practices were not statistically different during the pre-farm activities ($p > 0.05$). In contrast, for monsoon rice, the pre–farm emission in NPHR, CTHR and CTLR was similar ($p > 0.05$) but NPHR had 17 % lower emissions than CTLR ($p < 0.05$) (Figure 4).

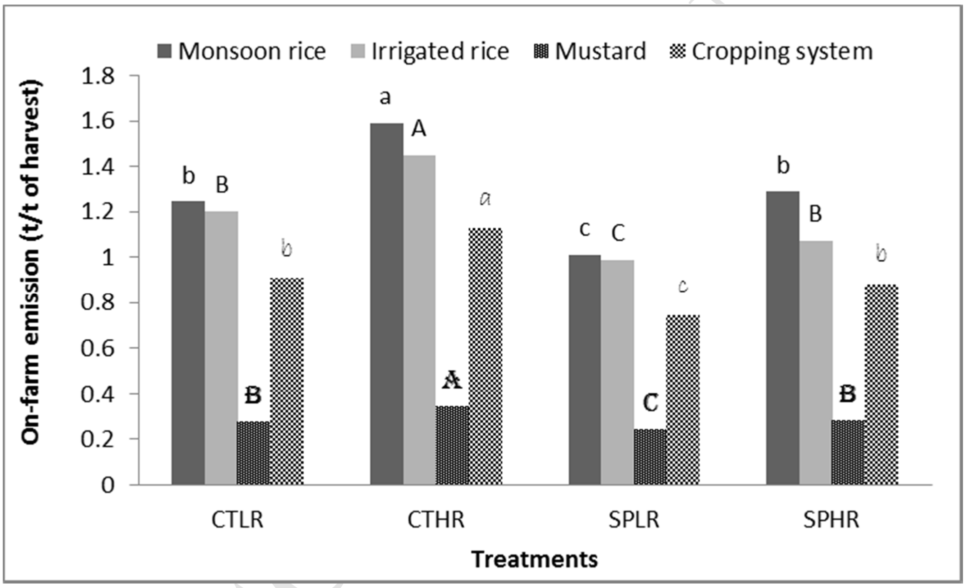


Figure 4. On-farm life cycle greenhouse gas emissions produced per season for one tonne of rice, or rice equivalent yield (REY) for mustard and for the cropping system production as influenced by crop establishment techniques and residue retention ($p<0.05$). Bars with the same letter above them are not significantly different at $p<0.05$. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

Overall, the pre–farm emissions of the cropping system life cycle GHG were very much lower ($p< 0.05$) than given off during the on–farm stage and comprised only 15.1, 11.9, 17.2 and 14.0 % of the cropping system life cycle GHG for CTLR, CTHR, NPLR and NPHR, respectively. Pre-farm emissions for mustard comprised twice as much relatively as the total on-farm emissions (29.8 % - 43.6 %) for monsoon rice (14-22 %) and four times as much (7.5-11.4 %) as for irrigated rice (Figure 3).

Irrespective of the component crops, NP/SP with LR and HR avoided 11 – 47 % of cropping system life cycle GHG emissions relative to CTLR and CTHR, respectively. To be specific, CA crop establishment practices with return of minimal residue (SPLR) cut down the on–farm emissions by the highest amount largely because it had the lowest emission of on-farm CH_4 emission. Specifically, SPLR reduced 51 %, 22 % and 18 % of on-farm emissions relative to CTHR, CTLR and SPHR, respectively, while SPHR reduced them by 28 % relative to CTHR and 3% relative to CTLR.

The on-farm emissions t^{-1} of grain production of rice were higher than the on-farm life cycle GHG t^{-1} of cropping system REY, while the on-farm emission t^{-1} of REY of mustard production was lower than the on-farm life cycle GHG t^{-1} of REY of cropping system. Among the crop establishment techniques, CTHR led the total cropping system life cycle GHG emissions from 1 tonne of REY production (Figure 4). However, NPLR categorically reduced CH_4 and thereby on–farm emissions. In both cases of rice production, total life cycle GHG emissions per tonne of rice production in NPHR exceeded NPLR even though NPHR outperformed NPLR in terms of yield. This is mainly because the greater CH_4 emissions in NPHR outbalanced the yield benefits of the increased residue retention. In a similar way, GHG emission for mustard cultivation showed that SPLR was the lowest contributor of GHG to the life cycle GHG (28 % and 11 % lower than CTHR and CTLR, respectively) for the production of a unit of REY of mustard ($p < 0.05$), followed by SPHR (Figure 4 and 5).

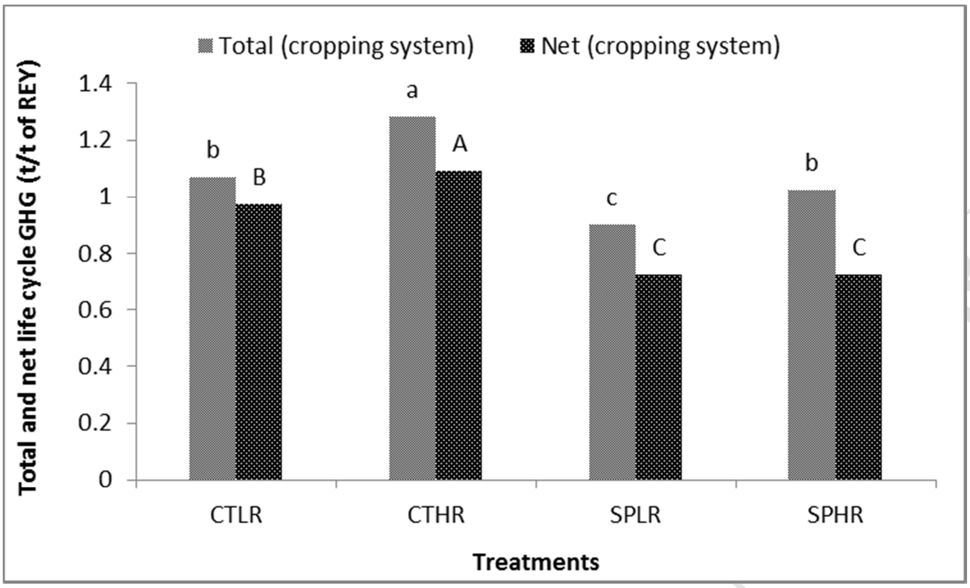


Figure 5. Average total and net life cycle greenhouse gas emitted for the production of one tonne of rice or rice equivalent yield in the rice-dominant cropping system as influenced by crop establishment techniques and residue retention. Net life cycle greenhouse gases emitted were calculated after subtracting the soil organic C sequestration from the total. Comparisons are made among emissions converted to CO_2eq according to global warming potentials of CO_2, CH_4 and N_2O over 100-year time horizons. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

3.2 GHG emissions within the LCA system boundary

Emission at pre–farm stage: On average, the pre-farm emissions tonne^{-1} REY of the rice-based cropping system were 0.161, 0.152, 0.156 and 0.143 tonne of CO_2eq for CTLR, CTHR, SPLR and SPHR, respectively ($p<0.05$). In contrast, for mustard crop cultivation, the life cycle GHG emissions associated with pre-farm logistics were 0.114, 0.104, 0.107 and 0.098

tonnes tonne^{-1} REY under CTLR, CTHR, SPLR and SPHR, respectively (p>0.05). Considering the cropping system life cycle GHG, monsoon rice, irrigated rice and mustard contributed 49 %, 28 % and 23 %, respectively of the pre-farm stage emissions. The relative contributions of treatments up to pre-farm stage were similar for monsoon rice, irrigated rice and mustard (Figure 4).

The NPHR had 17 %, 11 %, 9 % lower pre-farm emissions than CTLR, CTHR and NPHR, respectively (Figure 3). In case of monsoon rice production, the pesticide and fertilizer manufacturing accounted for 13 %, 10 %, 15 % and 11 % to the CO_2eq emissions of total GHG during the pre–farm stage for CTLR, CTHR, NPLR and NPHR, respectively. The production of inputs contributed 13 % in CTLR, 11 % in CTHR, 15 % in NPLR and 12 % in NPHR to the net life cycle GHG emissions within the pre–farm boundary (Figure 3). Again, for irrigated rice production, the pesticide and fertilizer manufacturing alone contributed 8% in NPLR, 6% in NPHR, 6% in CTHR and 6% in CTLR to the CO_2eq emissions of total GHG. On the whole, among the different activities, the manufacture and transport of inputs to paddock contributed the major shares, respectively. And among the different inputs, fertilizer provision up to paddock comprised the highest portion of the pre-farm emissions (Figure 3).

On–farm stage: Overall, on-farm processes of the rice–based cropping system comprised 83–88 % of the cropping system life cycle GHG, having the lowest portion with NPLR and the biggest portion with CTHR (Figure 4). Monsoon rice, irrigated rice and mustard crop growing under CT with LR and HR contributed 46 %, 44 % and 10 %, respectively, of the cropping system life cycle on-farm GHG, while the crops growing under SPLR and SPHR contributed 45, 44 and 11 % and 49, 40 and 11 %, respectively, for the three crops.

The greenhouse gas emissions from biogenic sources and farm machinery use ranged from 89 at NPLR to 93 % at CTHR and 78 at NPLR to 86 % at CTHR of GHG emissions during irrigated and monsoon rice production, respectively, while the emissions from RE mustard yield at on-farm stage were 70 % at CTLR to 77 % at CTHR. The GHGs emitted by CTLR practice were not different from NPHR/ SPHR ($p > 0.05$) for mustard and monsoon rice, in spite of keeping increased residue in the field (Figure 4). Conventional puddling with HR comprised the highest emissions during on-farm stage of irrigated rice production (Figure 4). In contrast, the SP/NPLR was most effective in saving GHG emissions compared to other tillage and residue retention combinations.

3.3 Overall GHG emissions

For the production of 1 tonne of irrigated rice, monsoon rice and mustard after accounting for soil sequestered C, net life cycle GHG emissions followed the sequence of NPLR/SPLR < NPHR/SPHR < CTLR < CTHR practices. Production of 1 tonne of REY of the rice–based cropping system caused 0.72, 0.72, 0.97 and 1.09 tonne of net CO_2eq life cycle GHG emission estimated under limited focused GWPI category, while the total life cycle GHGs for production of 1 tonne of REY were 0.9, 1.07, 1.11 and 1.3 t CO_2eq in SPLR, SPHR, CTLR and CTHR, respectively (Figures 2 & 5).

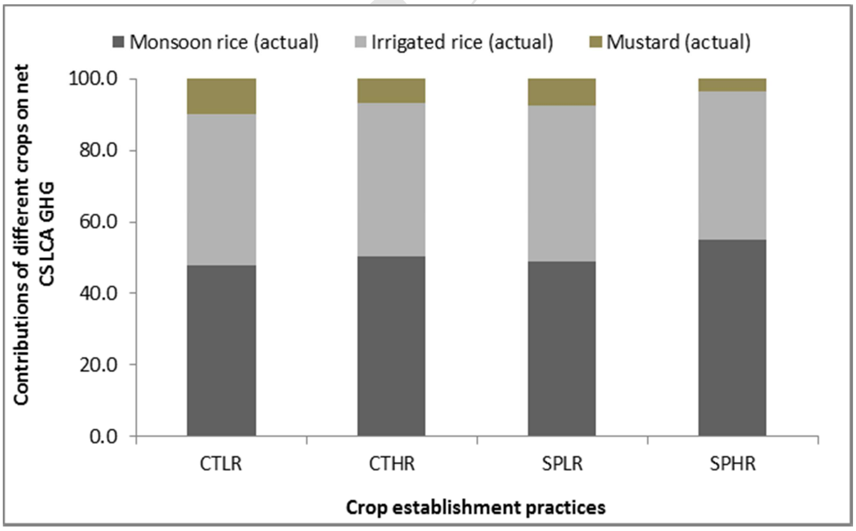


Figure 6. Relative contributions of component crops (%) to net cropping system life cycle greenhouse gas emission. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

3.4 Contributions of component crops to the CS LCA GHG

Relative contributions of component crops to the life cycle GHG of the rice-based cropping system varied due to different crop establishment and residue retention practices. Irrespective of crop establishment practices, the monsoon rice contributed the highest portion of the net cropping system life cycle GHG, followed by irrigated rice. For net cropping system life cycle GHG, monsoon rice accounted for 47.7-55.1 % of the estimated for rice-based cropping system of which the lowest portion was contributed by SPLR and the highest portion was by CTHR. The relative contribution of the irrigated rice was 41.4-43.7 % of the net cropping system life cycle GHG, while REY of mustard contributed 3.5-9.7 % of the net cropping system life cycle GHG. For mustard, SPHR contributed the lowest cropping system life cycle GHG and CTLR the highest GHG (Figure 6).

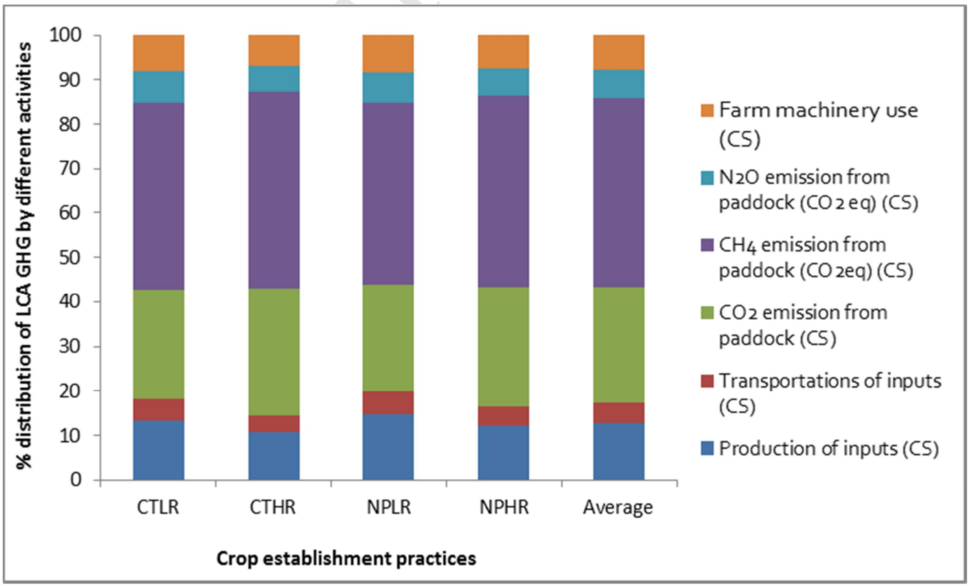


Figure 7. Greenhouse gas emissions (GHG) (%) based on life cycle analysis GHG contributed by different activities under different crop establishment and residue retention practices. [Legend: CT–Conventional puddled transplanting of rice; NP–Non-puddled transplanting of rice; LR–Low residue retention level; HR–Increased residue retention level]

3.5 Hotspots of the CS LCA GHG

The CH_4 emitted from both irrigated and monsoon seasons was the largest share of CO_2eq emissions by each practice, contributing 40.9 to 44.4 % to the total cropping system life cycle GHG under limited focus GWI estimation. The CO_2 emission from soil during the period of crop growth contributed 24.1 to 28.6% life cycle GHG emission in monsoon rice. The relative contribution of production of inputs comprised 10.6 to 14.9 % of the total cropping system life cycle GHG, followed by the emission by farm machinery use (6.8 to 8.2 %). The N_2O emissions made up only 5.8 to 7.2 % of the total life cycle GHGs (Figure 7). Transport of inputs contributed the lowest portion to the total cropping system life cycle GHG (3.8 to 5 % of the total emissions) (Figure 7).

By contrast with rice crops, the CO_2 emissions from upland mustard soil comprised the major portion (41 at SPLR to 66 % at CTHR) of life cycle GHG recorded under the four scenarios, while production of inputs (17.5–20.4 %) and N_2O emission from soil (14–15.7 %) were also important sources (Figure 7). Farm machinery use and transport of inputs accounted for 6.5–8.3% and 7.2–8.3%, respectively, of the total emissions from upland mustard soil (Figure 7).

4. Discussion

4.1 Cropping system GHG emissions of a rice-based intensive triple cropping system (CS LCA GHG)

Net life cycle GHG emissions estimated under the single impact GWI allocation for the rice-dominant cropping system were suppressed by minimum soil disturbance regardless of residue retention level. By contrast, minimum soil disturbance (SP or NP) with LR had the lowest total life cycle greenhouse gas (LCA GHGs) emissions for the rice-dominant cropping system (for one tonne of rice equivalent yield of the system) (Figure 5). Hence, for the cropping systems, as for a single crop (Alam et al. 2018), net life cycle GHG emissions need to be calculated because total life cycle GHG can give misleading conclusions about the overall carbon footprint as well as the impact of specific soil management approaches. Adopting less disturbance of soil for establishment of all three crops allowed up to 65 % greater SOC accumulation over five years of CA (Alam et al., 2018). The effect of the soil management technologies on SOC accumulation take several years to become manifest hence life cycle GHG for the cropping system is best calculated after the impacts of soil and crop management technologies have reached an equilibrium with the new level of SOC.

The CA practices caused less CH_4 emission under submerged rice soils and lower N_2O emission (Alam et al., 2016) during both rice and mustard seasons. The decreased soil disturbance may maintain lower soil microbial activities under upland mustard soil condition that limits heterotrophic microbial respiration (CO_2) and N_2O emissions. The decreased soil disturbance and non-puddled paddy field preparation for rice seedling establishment maintains higher redox potential under NP relative to puddling of soils following several tillage practices, which helps to reduce CH_4 emissions from rice growing soils (Alam et al., 2016; Alam et al., 2018). Lower standing water depth was also found under soils of non-puddled transplanting of rice (Alam et al., 2019). The Eh values ranged from around –200 mV to 300 mV at CT with LR and HR, while NP with LR and HR had a range of Eh values from –150 mV to –250 mV in soils under submerged condition (Alam et al., 2016 and Alam et al., 2019). The higher Eh values under CT with LR and HR might decrease CH_4 production by bacteria or increase CH_4 oxidation, resulting in diminished emission of CH_4 from submerged soil-rice arenchyma system (le Mer and Roger, 2001). Apart from these, the input requirements for all crops grown in the study under NP/SPLR and NP/SPHR were also lower relative to conventional practices. Soil temperatures are also generally lower under HR (Alam et al., 2018). Collectively, these reasons probably explain reduced emissions of life cycle GHGs from the rice-based cropping system under CA practices. On the other hand, the higher total and net life cycle GHGs under CTHR and CTLR practices can be attributed to heavy disturbance of soils by tillage (6 or more times per year) followed by puddling of soil (two times per year) which exacerbates the anaerobic conditions and resulted in a lower redox potential of soil (Alam et al., 2016). The anaerobic, saturated rice soil conditions created very rapidly after submergence (within hours) (Adhya et al., 2000; Bodelier, 2003) favour the increase of methanogenic bacteria populations and production of the by–product CH_4 through the anaerobic microbial respiration. The increased residue incorporation under CT of soils facilitates supply of substrate to methanogens and also stimulates the organisms to grow luxuriantly. Neue (1993) and Minamikawa et al. (2006) also reported that the application of carbon-rich straw helps methanogens survive and lowers redox potential in soils.

The sequestration of C was more in soils under SPHR over five years of CA cropping than other treatments (Alam et al., 2018). The increase in SOC can be attributed to: retention of residues as cover after each crop as well as the increase in C addition to the soil due to increase in biomass yield; minimal disturbance of SOM and plant root biomass; decreased CO_2 emissions and; the diversity of crops grown each producing different qualities of residues (Baldock, 2007; Alam et al., 2018). Hence, the lower methane emissions coupled with C sequestered in soils are primary reasons for the lower cropping system life cycle

GHGs (both total and net) for 1 tonne of REY under NP/SPLR and NP/SPHR practices (Figure 2, 4, 5 and 7).

Breiling et al. (1999) and Brodt et al. (2014) had also estimated only the GWI of rice paddy production, while Blengini and Busto (2009), Kasmaprapruet et al. (2009), Hokazono and Hayashi (2012) and Hokazono et al. (2009) prepared a multiple-impact life cycle GHG of paddy rice production. However, reports on the life cycle GHG for a complete cropping cycle are few, let alone estimates of the multiple-effects of environmental burdens.

4.2 Contributions of component crops to the CS LCA GHG

Though rice crops comprised the major part of pre-farm emissions of cropping system life cycle GHG, the emissions at the pre-farm stage of production for crops of the rice-dominant cropping system were significantly lower than similar LCA studies involving single crops of rice (Wang et al., 2010; Thanawong et al., 2014; Alam et al., 2016; Yadav et al., 2018). However, estimation of GHG emissions through the LCA approach in a rice-upland cropping system has not been reported until now. The present study grew high yielding varieties with low rates of fertilisers (because of the adequate soil nutrient status) and used very minimum levels of fungicides, insecticides and herbicides (as crop species apart from monsoon rice were changed every 2-3 years to break disease and weed cycles) which collectively explain the low pre-farm life cycle GHG emissions. More importantly, use of low GHG emitting raw material (feed–stock) (i.e. natural gas) for producing urea fertiliser and generating electricity and light vehicle use for input transport to crop fields in the experimental region (Alipur, EGP) are behind the low pre-farm cropping system life cycle GHG emissions (Alam et al., 2016; Alam et al., 2019). In addition, the improvement of soil fertility status due to adopting crop rotation and practicing CA over five years created co-benefits in terms of yield increase and SOM increase (Hokazono and Hayashi, 2015; Alam et al., 2018).

The relatively lower contribution of mustard (after conversion to REY) relative to rice crops can be attributed to lower input requirements especially N which minimises the N_2O emissions (Figure 5). Methane, the major GHG emitted during rice cultivation under wetland condition was absent under upland mustard condition. Among the component crops, the mustard crop requires only supplemental irrigation twice in a season as the variety selected for the study was a very short-duration crop (mustard, *cv.* BARI mustard-14) (Azad et al., 2017).

The higher relative contribution of monsoon rice (49 % vs 28 and 23 % for irrigated rice and mustard, respectively) to cropping system life cycle GHG can be attributed to its low yield. In the present study, yields of irrigated rice ranged from 6.2 in CTLR to 6.7 t ha^{-1} in NPHR, whereas, in monsoon season, the yield ranged from 4.6 in CTLR to 5.3 t ha^{-1} in NPHR (Table 1; Alam et al., 2018). On the other hand, the REY of mustard ranged from 7.32 in CTLR to 8.42 t ha^{-1} in NPHR (Table 1). The production of REY of mustard used very low amounts of inputs but had high REY when compared with monsoon rice production which needed more carbon intensive inputs for lower yields. Furthermore, the life cycle GHG in the pre-farm stage in mustard cultivation was 45-48% lower than pre-farm emission of monsoon rice production per tonne. Brodt et al. (2014) reported rice grain yield of 9.3 Mt ha^{-1}, whereas Wang et al. (2010) reported rice yield (8.8 Mt ha^{-1}), but the pre-farm emission was lower in the former case because the latter case was input–intensive and used more than double the inputs. Fusi et al. (2014) also reported that input (mostly fertilisers), distribution and input usage t^{-1} harvest amounted to 30–40 % of the life cycle GHG. In studies of sunflower and rapeseed oil production, Badey et al. (2013) found pre-farm emission shared more than 50 % of the emission up to seed production. In the current study, the pre-farm emissions for inputs of rice crop production in both irrigated and monsoon season were similar to studies of Badey et al. (2013), Xu et al. (2013) and Blengini and Busto (2009). But the pre-farm emissions for rice production in the current study are higher than the study of Thanawong et al. (2014) and Wang et al. (2010). The economically valuable crop mustard in the rice-based system comprised only 4-11 % of net cropping system life cycle GHG. The mustard crop contributed only 23% to the pre-farm emissions per tonne of REY production which reduced the cropping system life cycle GHG to a great extent (Figure 5). From studies conducted on grain crops under upland conditions, Nemecek et al. (2015) and Gan et al. (2011) found that growing diversified crops in rotations can assist in reducing the average C footprint per crop. The component crops in the rotation determine not only the crop and environmental performances, but also inputs of fertilizer (especially N), provision of mechanisation and use of other chemicals (Crozat and Fustec, 2004; Deike et al., 2008). Nemecek et al. (2015) also found that diverse crops in the cropping systems with reduced chemicals use (fertilizer, pesticide etc.) are promising means to curb the environmental impacts of intensive arable cropping systems. Lemke et al. (2007) suggested that inclusion of pulses in cereals-based cropping systems in the Northern Great Plains (NGP) region influenced the balance of the systems' net GHG as the multi-crop systems required variable pesticides and fertilizers and had residues of varied quality and quantity compared with cereals only cropping systems. On the other hand, Burton et al. (2008) found that increased fertilizer N application as required for potato and single time of application commonly increases N_2O emissions (Ruser et al., 2001; Zebarth et al., 2008) and, that, N_2O production increases non-linearly with increased N fertilizer (McSwiney and Robertson, 2005).

Alam et al. (2016) found that life cycle GHG generated by consumed fuel during preparing and irrigating land and harvesting made up 14 to 19 % of the on-farm stage life cycle GHG of irrigated rice. That irrigating rice crop shared major part of energy required for on-farm activities is confirmed by other studies of Islam et al. (2013) and Khan et al. (2009). On the contrary, the monsoon rice in the present study did not require any irrigation application and saved those life cycle GHGs. Selecting suitable crops and growing them in rotation can have favourable effects on GHG emissions (Dukes, 2003).

The present study contrasted with the study by Thanawong et al. (2014) who recorded increased on-farm GHG emission from soils of irrigated rice, relative to rain-fed rice. They attributed this increased on-farm emissions to augmented synthesis and release of CH_4 from continuous submerged soil of irrigated rice (the CH_4 synthesised and emitted in irrigated rice soil was twice the amount of CH_4 emitted from rain-fed rice soil). The continuous ponding of water in the irrigated rice field might have create more reduced soils and thereby caused increase CH_4 emission. Irrigation system installation and irrigating crops also accounted for additional emissions to the total emissions.

4.3 Identification of hotspots

In the rice-upland rotation, on–farm CH_4 emissions contributed the highest overall emissions, followed by CO_2 emissions from soils due to heterotrophic respiration, input manufacturing and delivery, the emission by farm machinery use and N_2O emissions from fields (Figure 7). By contrast, N_2O dominated GHGs recorded in C footprint studies of upland (arable) crops (Grant and Beer, 2008; Gan et al. 2012b; Eshun et al., 2013; Weller et al., 2014). Episodes of alternative aerobic and anaerobic (waterlogging) conditions increase N_2O emission (Flessa and Beese, 1995), while CH_4 becomes oxidised by microbes under aerobic conditions of soil, resulting in negative emission of CH_4 (Gilbert and Frenzel, 1998; Ettwig et al., 2010). The experiment results observed by Kritee et al. (2018) suggested that the Indian subcontinent's N_2O emissions from intermittently flooded rice fields could be 30–45 times higher than reported under continuous flooding. On the other hand, the present study had two rice crops; one was maintained with continuous flooding by irrigation, the other was rainfed but continuously flooded by natural rainfall events (Alam et al., 2019). Under the present conditions, little of the fertilizer-derived NH_4-N would be oxidized biologically to NO_3-N in anaerobic soil conditions which would lower the risk of NO_3-N leaching and N_2O production due to denitrification (Savant and De Datta, 1982). Zhang et al. (2017) surveyed the C footprints of maize, wheat and rice across China and concluded that the use of N containing fertilisers, straw burning, energy in farm machinery use and irrigation, and CH_4 emanating from wetland rice soils are the factors contributing to total C-eq: their shares range from 8 to 49 %, 0 to 70 %, 6 to 40 %, 0 to 44 % and 15 to 73 %, respectively. Alam et al. (2016) in their LCA of irrigated rice in the EGP–Bangladesh and Blengini and Busto (2009) who studied the C footprint of rice production through a LCA approach in Italy identified similar hotspots for the production of rice crops (CH_4 emissions from anaerobic soil, emissions from fuel use for machinery operation, and provision of fertilisers). A hotspot from CO_2 emissions from soils under mustard can be attributed to heterotrophic respiration in the aerobic soil during the cool-dry season. As the soil had improved nutrient status, input of fertilizers for mustard was minimal. Hence, CO_2 emission was the main emitter of life cycle GHGs for mustard. In a study of monsoon rice LCA (Alam et al., 2019), CH_4 emission from on-farm stage production comprised the majority of the life cycle GHGs.

Globally, the IPCC (2007) attributed 55 % of agriculture-generated global CH_4 flux to wetland rice. Alam et al. (2016), Bacenetti et al. (2015), Harada et al. (2007) and Pathak et al. (2005) estimated contributions of CH_4 to the life cycle GHGs of rice crop to be the principal hotspot which made up around 60 % of total rice life cycle GHG. Similarly, Fumoto et al. (2008), Hokazono and Hayashi (2012) and Hatcho et al. (2012) assessed rice in waterlogged soil in Japan and concluded that CH_4 was the main contributor to life cycle GHG. A life cycle GHG of rice in France also identified CH_4 as the topmost contributor to GWP (Drocourt et al., 2012). Though the share of biogenic CH_4 from production of rice under wetland condition in Italy (Milano, Pavia, Vercelli and Novara areas) represented only 40 % of the life cycle GHG, the CH_4 attributed to organic matter decomposition was the highest contributor to GWP (Fusi et al., 2014). The present study identified CH_4 as the major hotspot for GHG emissions both in rice crops and in the cropping system, but showed that the minimum soil disturbance practice, including non-puddled transplanting, was effective in decreasing methane emission although this gain was offset to some extent by the higher residue retention. This suggests that CA practices have the potential to decrease the C footprint of cropping systems based on wetland rice. There may be further opportunities to modify N fertilising tactics and increase N use efficiency to reduce the net emissions and C footprint of wetland rice crops by almost one-third to half in CA production technologies for rice-based cropping.

4.4 Overall emissions of GHG

After accounting for sequestrated C in soil, net life cycle GHGs produced t^{-1} of REY by the cropping system under the GHG intensity allocation amounted to 0.73, 0.74, 0.98 and 1.12 tonne for CTLR, CTHR, SPLR and SPHR, respectively. By contrast, in China, GHGs produced were 5.9 t CO_2eq t^{-1} rice versus 1.76 t CO_2eq t^{-1} maize and 2.75 t CO_2eq t^{-1} wheat (Zhang et al., 2017). The footprints in the current study for production of upland mustard or wetland rice either by irrigation or by monsoon rain are lower than footprints found in China (Zhang et al., 2017). Hokazono and others (2009) on puddled rice soil in Japan found life cycle GHG emissions within the cradle to farm-gate boundary were 1.62 t CO_2eq in organic farming, 1.34 t CO_2eq in sustainable farming and 1.51 t CO_2eq in conventional farming t^{-1} production of rice, respectively. The life cycle GHG estimated by Farag et al. (2013) from cradle to farm-gate boundary was 1.9 t CO_2eq t^{-1} rice, while the puddled transplanted rice accounted for 2.21 t life cycle GHG (CO_2eq t^{-1} rice) within the system boundary up to farm-gate boundary (Ryu et al., 2013). Hokazono and Hayashi (2015) found that growing non rice crops or legume crops (capable of N fixation) requiring less inputs in place of continuous rice tended to be efficient in mitigating life cycle GHGs.

In addition to the emissions recorded for the cultivation of mustard, irrigated rice and monsoon rice in rotation in the EGP during their growing cycle, the fallow periods during the transition from upland crops to wetland rice and from wetland crops to upland winter crops might cause additional emissions of GHGs (CH_4, CO_2 and N_2O). Sander et al. (2018) found during drying of soils, that emission of N_2O started following the accumulation of NO_3-N. On the other hand, when NO_3-N started disappearing during flooding of soils during land preparation, NH_4-N started accumulating with the mineralised N. However, among the GHGs, N_2O is emitted through nitrification and denitrification processes during the periods of transition (Sander et al., 2018). The emissions of CH_4 and CO_2 occur throughout the growing season and in the periods of fallow between crops. Hence the present findings should be extended to LCA studies that include emissions during the fallow periods between crops (Martínez-Eixarch et al., 2018).

4.5 Implications for cleaner production

The LCA estimated in the present research suggests a GHG mitigation and cleaner production strategy for production of crops in rice-based cropping systems. Out of five cleaner production strategies, including input substitution, good housekeeping, product modification, technology modification and on-site recycling, the present research considered technology modification (e.g. strip tillage for upland crops and non-puddling of rice crops) and good housekeeping (e.g. residue retention) strategies to reduce GHG mitigation. In rice-based systems of the EGP, a range of upland crops are grown in the cool-dry season (from mid-October to middle March). Some crops (e.g. potato) require high inputs which may lead to increased GHG emissions from N containing fertilisers, irrigation etc. The LCA for GHG emissions of rice crops need to be combined with those for the upland crops to complete LCAs of the cropping systems with diversified crops that are typical of the EGP. Indeed, the present study highlights the importance of regional factors such as the light machinery use and gas generation of electricity, both of which are relevant to Bangladesh and the EGP, and distinguish the life cycle GHG for this region from other regions. The inventory and EFs developed in the present study will be useful for further life cycle GHG studies in the crop production sector in the EGP. Conservation agricultural practices have been reported to increase SOC in some studies (Alam et al. 2018; Salahin, 2017; West and Post, 2002), but not in others (Powlson et al., 2016). Where soil and crop management practices increase sequestered soil C, inclusion of the SOC gains in the LCA inventory will improve accuracy of the LCA tool for determining the net GHG values per functional unit of REY production. This would enable policy makers to accurately predict the benefits of CA practices for GWP mitigation. The present study which estimated C footprints of a rice-based cropping system can inform policy development by Governments in the EGP since wetland rice is the dominant crop in the region and a major contributor to national carbon accounts. The methodology followed for estimating C footprints of CA cropping could be used for countries growing crops following CA principles.

5. Conclusions

The C footprint of intensive rice-based cropping system has been estimated under crop establishment practices following CA principles for all three component crops in rice-based cropping system in the EGP. Strip planting for upland crops and non-puddling of rice crops with minimal soil disturbance and increased residue retention were identified as two alternative crop management practices for cleaner production strategies than the conventional production system of crops in rice-intensive systems. The CA cropping techniques save fuel, improve productivity, reduce GHG emissions and thereby offers environmental benefits. The estimation of the C footprint of grain production in the EGP has been improved by accounting for SOC-sequestration in the life cycle GHG analysis. Since SOC varies with cropping and soil management practices as well as among regions, accounting for SOC can increase the accuracy of carbon footprint accounting or GWI calculation.

The minimum soil disturbance crop establishment practice for upland crop (SP) and for irrigated and monsoon rice (NP) with current or increased residue inputs offer significant GHG savings at both pre-farm and on-farm stages of component crop production of the rice dominant cropping system relative to conventional methods of rice crop establishment in the EGP. Increased SOC sequestration and the reduced GHG emission by following minimum disturbance of soil, residue return management and crop rotation, the key principles of CA, collectively reduced life cycle GHG of the cropping system compared to conventional tillage. For net life cycle GHGs, strip planting/non-puddled transplanting with minimal or high residue return was equally efficient choice of GWP mitigation. The CTLR and CTHR accounted for 0.73 and 0.74 tonne net life cycle GHGs for one tonne REY of the rice-dominant cropping system. The savings with the best mitigation practice (NP/SPLR and NP/SPHR) for net life cycle GHGs were 0.37 and 0.25 t emissions t^{-1} of rice production relative to CTHR and CTLR, respectively.

The on–farm stage of the rice-based cropping system contributed 83 (SPLR)–88% (CTHR) of the cropping system life cycle GHG, while for monsoon rice it contributed 78 % (NPLR) to 86 % (CTHR) of the life cycle GHG emissions. For RE mustard yield, the on-farm stage contributed 70 % (CTLR) to 77 % at CTHR of the total. These on-farm emissions were due predominantly to emission of greenhouse gases from cropped soil and to emissions from fuel use for usage of machineries. Regardless of crop establishment practices and residue return, CH_4 was the dominant GHG emitted t^{-1} of irrigated and monsoon rice production in the EGP due to soil submergence (anaerobic) maintained for rice crop production, while for mustard, CO_2-eq emission from on-farm stage was the predominant GHG. Carbon dioxide emission from soil, emissions associated with fuel use for on-farm machineries as well as production of inputs were significant contributors to life cycle GHGs of monsoon rice production, irrigated rice production and REY of mustard, respectively. Total life cycle GHG emissions overestimate the C footprint in the long term when significant SOC differences emerge among cropping and soil management practices: net life cycle GHG which account for changes in soil sequestered C should be determined in these cases. Further modifications of the management practices for component crops of the intensive triple-crop system that lead to yield increase or decreased inputs could further improve the net life cycle GHG performances of the CA practice.

6. Acknowledgements

We acknowledge the analytical service laboratories provided by the Division of Soil Science, Bangladesh Agricultural Research Institute and Bangladesh Rice Research Institute at Gazipur. The authors are grateful to the farmer who permitted the long term use of their land for conducting the experiment. The Institute of Fuel Research and Development (IFRD) and Soil, Agronomy and Environment Section of the Biological Research Division of Bangladesh Council of Scientific and Industrial Research kindly provided analytical support. We are thankful to ACIAR (Project LWR 2010/080) for funding support.

7. References

Abbas, D. 2011. Energy use efficiency and economic analysis of canola production in three different areas in Iran. ARPN J. Agric. Biol. Sci. 6(11), 54–61.

Adhya, T.K., Bharati, K., Mohanty, S.R., Ramakrishnan, B., Rao, V.R., Sethunathan, N., Wassmann, R., 2000. Methane emission from rice fields at Cuttack, India, Nutr. Cycling Agroecosys. 58, 95–105.

AIJ (Architectural Institute of Japan), 2003. LCA database based on inter–industry relations table in 1995. In: Guiding Principle of LCA for Architectures, Tokyo: Architectural Institute of Japan. Available from URL: http://news–sv.aij.or.jp/tkankyo/s0/news.htm. (Accessed on 09 November 2014)

Alam, M.K., Biswas, W.K., Bell, R.W., 2016. Greenhouse gas implications of novel and conventional rice production technologies in the Eastern-Gangetic plains. J. Clean. Prod. 112, 3977–3987.

Alam, M.K., Bell, R.W., Haque, M.E., Kader, M.A., 2018. Minimal soil disturbance and increased residue retention increase soil carbon in rice-based cropping systems on the Eastern Gangetic Plain. Soil Till. Res. 183, 28-41.

Alam, M.K., Biswas, W.K., Bell, R.W., 2019. Increases in soil sequestered carbon under conservation agriculture cropping decrease the estimated greenhouse gas emissions of wetland rice using life cycle assessment. J. Clean. Prod. In Press.

Anderson, J.P.E., 1982. Soil respiration. In Page A.L., Miller, R.H., Keeney, D.R. (Eds), Methods of soil analysis. Part 2. Agronomy Monograph, 9 (2nd ed., pp 831–871). Madison, WI: ASA and SSSA.

Anjeneyul, V.R., Singh, S.P., Paul, M., 1982. Effect of competition free periods and techniques and pattern of pearl millet planting on growth and yield of mungbean inter-cropping systems. Ind. J. Agronomy 27, 219-226.

Azad, A.K., Goswami, B.K., Rahman, M.L., Malakar, P.K., Hassan, M.S., Rahman, M.H.H., 2017. Edited KRISHI PROJUKTI HATBOI (Handbook on Agro-technology), 7th edition, Bangladesh Agricultural Research Institute, Gazipur 1701, Bangladesh (in Bangla).

Bacenetti, J., Fusi, A., Negri, M., Fiala, M., 2015. Impact of cropping system and soil tillage on environmental performance of cereal silage productions. J. Clean. Prod. 86, 49–59.

Badey, L., Lahitte, N., Flenet, F., Bosque, F., 2013. French environmental communication on sunflower and rapeseed oils based on life cycle assessment. OCL 20(4) A401.

Baldock, J.A., 2007. Composition and cycling of organic carbon in soil. In Marschner, P., Rengel, Z. (Eds.), Nutrient cycling in terrestrial ecosystems (pp 1–35). Berlin Heidelberg: Springer.

Bangladesh Business News, 2013. Bangladesh to import 50,000 tonnes of fertilizer from Belarus. Accessed on 6 September 2014. http://www.businessnews–bd.com/index.php?option=com_content&view=article&id=6209:bangladesh–to–import–50000–tones–of–fertilizer–from–belarus&catid=36:business&Itemid=27 (accessed 15 July 2014).

Barton, L., Thamo, T., Engelbrecht, D., Biswas, W.K., 2014. Does growing grain legumes or applying lime cost effectively lower greenhouse gas emissions from wheat production in a semi–arid climate? J. Clean. Prod. 83, 194–203.

BBS, 2013. Year Book of Agricultural Statistics of Bangladesh.Ministry of Planning, Government of the Peoples' Republic of Bangladesh, Dhaka.

Biswas, W.K., Barton, L., Carter, D., 2008. Global warming potential of wheat production in Western Australia: a life cycle assessment. Water Environ. J. 22, 206–216

Blengini, G.A., Busto, M., 2009. The life cycle of rice: LCA of alternative agri–food chain management systems in Vercelli (Italy). J. Environ. Manage. 90, 1512–22.

Bodelier, P.L.E., 2003. Interactions between oxygen releasing roots and microbial processes in flooded soils and sediments, In: de Kroon, H.,Visser, E.J.W., eds., Root Ecology. Springer, Berlin. Pp.331–362.

Bosch, P., Kuenen, J., 2009. TNO report on Greenhouse gas efficiency of industrial activities for European Commission, DG Environment, TNO Built Environment and Geosciences, Princetonlaan 6, the Netherlands, pp75

Brander, M., Sood, A., Wylie, C., Haughton, A., Lovell, J., 2011. Electricity–specific emission factors for grid electricity. Ecometrica 82, 1–22.

Breiling, M., Tatsuo, H., Matsuhashi, R., 1999. Contributions of rice production to Japanese greenhouse gas emissions applying life cycle assessment as a methodology. Tokyo, Laboratory for Land Resource Sciences, Department of Biological and Environmental Engineering, Graduate School for Agriculture and Life Sciences, The University of Tokyo, 32. http://www.breiling.org/publ/ lcaricejap-en.pdf. Accessed 25 May 2017

Brentrup, F., 2009. The impact of mineral fertilizers on the carbon footprint of crop production. The Proceedings of the International Plant Nutrition Colloquium XVI UC Davies, USA, 8 p

Brodt, S., Kendall, A., Mohammadi, Y., Arslan, A., Yuan, J., Lee, I. Linquist, B., 2014. Life cycle greenhouse gas emissions in California rice production. Field Crops Res. 169, 89–98

Burton, D.L., Zebarth, B.J., Gillam, K.M., MacLeod, J.A., 2008. Effect of split application of fertilizer nitrogen on N_2O emissions from potatoes. Can. J. Soil Sci. 88, 229-239.

Crozat, Y., Fustec, J., 2004. Assessing the role of grain legumes in crop rotation: some agronomic concepts that can help. In: AEP (Ed.), Grain legumes and the environment: how to assess benefits and impacts. AEP & FAL, Zurich, pp. 55–60, 18.–19.11.2004.

DEFRA, 2008. 2012 Guidelines to DEFRA/DECC's GHG Conversion Factors for Company Reporting, Produced by AEA for the Department of Energy and Climate Change (DECC) and the Department for Environment, Food and Rural Affairs, 17 Smith Square, London, Version–1

Deike, S., Pallutt, B., Melander, B., Strassemeyer, J., Christen, O., 2008. Long-term productivity and environmental effects of arable farming as affected by crop rotation, soil tillage intensity and strategy of pesticide use: a case-study of two long-term field experiments in Germany and Denmark. Eur. J. Agron. 29, 191–199.

Denham, F., Biswas, W., Solah, V., Howieson, J., 2016. Greenhouse gas emissions from a Western Australian finfish supply chain. J. Clean. Prod. 112, 2079-2087.

Dobbie, K.E., Smith, K.A., 2006. The effect of water table depth on emissions of N_2O from a grassland soil. Soil Use Manage. 22, 22–2810.

Drocourt, A., Mervant, Y., Milhau, F., Chinal, M., Hélias, A., 2012.Environmental assessment of rice production in Camargue, France. In: Corson, M.S., van der Werf, H.M.G. (Eds.), Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2012), 1-4 October 2012, Saint Malo, France. INRA, Rennes, France, p. 824-825.

Dukes, J.S., 2003. Burning buried sunshine: Human consumption of ancient solar energy. Climatic Change 61, 31–44.

Duxbury, J.M., Harper, L.A., Mosier, A.R., 1993. Contributions of agroecosystems to global climate change. In: Agroecosystem effects on radiatively important trace gases and global climate change. Harper, L., Mosier, A.R., Duxbury, J.M., Rolston, D.E., (eds) 1–18. ASA Special Publication no:55. Amer. Soc. Agron. Madison. WI.

Ellert, B.H., Bettany, J.R., 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Can. J. Soil Sci. 75, 529–538

Eshun, J.F., Apori, S.O., Wereko, E., 2013. Greenhouse gaseous emission and energy analysis in rice production systems in Ghana. African Crop Science J. 21, 119 – 125

Ettwig, K.F., Butler, M.K., Le Paslier, D., Pelletier, E., Mangenot, S., Kuypers, M.M., et al., 2010. Nitrite driven anaerobic methane oxidation by oxygenic bacteria. Nature 464, 543-548.

Farag, A.A., Radwan, H.A., Abdrabbo, M.A.A., Heggi, M.A.M., McCarl, B.A., 2013. Carbon footprint for paddy rice production in Egypt. Nat. Sci.11, 36–45.

Finkbeiner, M., Tan, R., Reginald, M., 2011. Life cycle assessment (ISO 14040/44) as basis for environmental declarations and carbon footprint of products, ISO Technical Committee 207 Workshop, Norway.

Flessa, H., Beese, F., 1995. Effects of sugarbeet residue on soil redox potential and nitrous oxide emission. Soil Sci. Soc. Am. J. 59, 1044–1051.

Freibauer, A., Rounsevell, M.D.A., Smith, P., Verhagen, J., 2004. Carbon sequestration in agricultural soils of Europe. Geoderma 122, 1–23.

Fumoto, T., Kobayashi, T., Li, C., Yagi, K., Hasegawa, T., 2008. Revising a process–based biogeochemistry model (DNDC) to simulate methane emission from rice paddy fields under various residue management and fertilizer regimes. Glob. Change Biol. 14, 382–402.

Fusi, A., Bacenetti, J., González–García, S., Vercesi, A., Bocchi, S., Fiala, M., 2014. Environmental profile of paddy rice cultivation with different straw management. Sci. Total Environ. 494–495, 119–128

Gan, Y., Liang, C., Campbell, C.A., Zentner, R.P., Lemke, R.L., Wang, H., Yang, C., 2012a. Carbon footprint of spring wheat in response to fallow frequency and soil carbon changes over 25 years on the semiarid Canadian prairie. Eur. J. Agron. 43, 175–184.

Gan, Y., Liang, C., Wang, X., McConkey, B., 2011. Lowering carbon footprint of durum wheat by diversifying cropping systems. Field Crops Res. 122, 199–206.

Gan, Y., Liang, L., Huang, G., Malhi, S.S., Brandt, S.A., Katepa-Mupondwa, F., 2012b. Carbon footprint of canola and mustard is a function of the rate of N fertilizer. Int. J. Lile Cycle Assess. 17(1), 58-68

Ghimire, R., Lamichhane, S., Acharya, B.S., Bista, P., Sainju, U.M., 2017. Tillage, crop residue, and nutrient management effects on soil organic carbon in rice-based cropping systems: A review. J. Integr. Agric. 16(1), 1–15

Gilbert, B., Frenzel, P., 1998. Rice roots and CH_4 oxidation: the activity of bacteria, their distribution and the microenvironment. Soil. Biol. Biochem. 14, 1903-1916.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39.

Grant, T., Beer T., 2008. Life cycle assessment of greenhouse gas emissions from irrigated maize and their significance in the value chain. Aust. J. Exp. Agr. 48, 375–381.

Haque, M.E., Bell, R.W., Islam, M.A., Rahman, M.A., 2016. Minimum tillage unpuddled transplanting: An alternative crop establishment strategy for rice in conservation agriculture cropping systems. Field Crops Res. 185, 31–39.

Harada, H., Kobayashi, H., Shindo, H., 2007. Reduction in greenhouse gas emissions by no-tilling rice cultivation in Hachirogata polder, northern Japan: Life–cycle inventory analysis. Soil Sci. Plant Nutr. 53, 668 – 677.

Hatcho, N., Matsuno, Y., Kochi, K., Nishishita, K., 2012. Assessment of environment–friendly rice farming through life cycle assessment (LCA). CMU. J. Nat. Sci. 11, 1–10. [Special Issue on Agr. Nat. Res.].

HBEFA, 2014. Handbook on emission factors of road transport (HBEFA).Version HBEFA 3.2. INFRAS, Graz

Heidari, M.D., Omid, M., 2011. Energy use patterns and econometric models of major greenhouse vegetable productions in Iran. Energy 36, 220–225.

Hokazono, S., Hayashi, K., 2015. Life cycle assessment of organic paddy rotation systems using land- and product-based indicators: A case study in Japan. Int. J. Life Cycle Assess. 20, 1061–1075.

Hokazono, S., Hayashi, K. 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. J. Clean. Prod. 28, 101–112

Hokazono, S., Hayashi, K., Sato, M., 2009. Potentialities of organic and sustainable rice production in Japan from a life cycle perspective. Agronomy Research 7(Special issue I), 257–262

Hutchinson, G.L., Livingston, G.P., 1993. Use of chamber system to measure trace gas fluxes, in: Rolston, D.E., Duxbury, J.M., Harper, L.A. Mosier, A.R.(eds), Agricultural Ecosystem Effects on Trace Gases and Global Climate Change, ASA Special Publications 55, American Society of Agronomy, Crop Science Society of America and Soil Science Society of America, Madison, USA, pp. 63–78.

IEA, 2012. Energy Technology Perspectives, 2012, Pathways to a Clean Energy System. IEA, Paris, France. p.329.

IFA, 2009. Fertilizers, Climate Change and Enhancing Agricultural Productivity Sustainably. IEA, Paris, France.

INFRAS, 2010. The Handbook Emission Factors for Road Transport (HBEFA), version 3.1. INFRAS in cooperation with further editors: FOEN/Switzerland; Umweltbundesamt

Dessau/Germany; Umweltbundesamt Wien/Austria; Swedish Road Administration, ADEME/France; SFT/Norway. Bern. http://www.hbefa.net/e/index.html

Intergovernmental Panel on Climate Change (IPCC). (2007). Agriculture. In B. Metz, O. R. Davidson, P. R. Bosch, & P. R. Bosch (Eds.), Climate Change 2007: Mitigation, Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom/New York, NY, USA: Cambridge University Press.

IPCC, 2006. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES, Tokyo, Japan.

IPCC, 2013. In: Stocker, T.F., Qin, D., Plattner, G.K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis, Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, p. 1535.

Iqbal, T., 2007. Energy input and output for production of boro rice in Bangladesh. EJEAFChe, Electron. J. Environ. Agric. Food Chem. 6, 2144–2149.

Islam, A.K.M.S., Hossain, M.M., Saleque, M.A., Rabbani, M.A., Sarker, R.I., 2013. Energy consumption in unpuddled transplanting of wet season rice cultivation in North West region of Bangladesh. Progress. Agric. 24, 229 – 237.

Islam, M.A., 2017. Conservation Agriculture: Its effects on crop and soil in rice-based cropping systems in Bangladesh, PhD thesis. School of Veterinary and Life Sciences, Murdoch University, Australia.

ISO, 2006. ISO 14044:2006. International Organization of Standardization. Geneva, Switzerland.

Jackson, M. L., 1973. Soil Chemical Analysis. Prentice Hall of India (pvt). Ltd. New Delhi.

Jantalia, C.P., dos Santos, H.P., Urquiaga, S., Boddey, R.M., Alves, B.J.R., 2008. Fluxes of nitrous oxide from soil under different crop rotations and tillage systems in the South of Brazil. Nutr. Cycl. Agroecosyst. 82, 161–173.

Kasmaprapruet, S., Paengjuntuek, W., Saikhwan, P., Phungrassami, H., 2009. Life cycle assessment of milled rice production: Case study in Thailand. Eur. J. Scient. Res. 3, 195–203.

Khan, S., Khan, M.A., Hanjra, M.A., Mu, J., 2009. Pathways to reduce the environmental footprints of water and energy inputs in food production. Food Policy 34, 141–149

Kritee, K., Nair, D., Zavala-Araiza, D., Proville, J., Rudek, J., Adhya, T.K., Loecke, T., Esteves, T., Balireddygari, S., Dava, O., Ram, K., Abhilash, S.R., Madasamy, M., Dokka, R.V., Anandaraj, D., Athiyaman, D., Reddy, M., Ahuja, R., Hamburg, S.P., 2018. High nitrous oxide fluxes from rice indicate the need to manage water for both long- and short-term climate impacts. PNAS 115(39), 9720–9725.

Lal, B., Panda, B.B., Gautam, P., Raja, R., Singh,T., Mohanty, S., Shahid, M., Tripathi, R., Kumar, A., Nayak, A.K., 2015. Input–output energy analysis of rainfed rice-based cropping systems in Eastern India. Agron. J. 107(5), 1750-1756

Lal, R., 2004. Carbon emissions from farm operations. Environ. Int. 30,981–990.

Lal, R., Follett, R.F., Kimble, J.M., Cole, C.V., 1999. Managing U.S. cropland to sequester carbon in soil. J. Soil Water Conserv. 54, 374–381.

Le Mer, J.L., Roger, P., 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol., 37, 25-50.

Lemke, R.L., Zhong, Z., Campbell, C.A., Zentner, R., 2007. Can pulse crops play a role in mitigating greenhouse gases from North American agriculture? Agron. J. 99, 1719–1725.

Liu, C., Cutforth H., Chai Q., Gan Y., 2016. Farming tactics to reduce the carbon footprint of crop cultivation in semiarid areas. A review. Agron. Sust. Dev. 36, 69-85.

Marble, S.C., Prior, S.A., Runion, G.B., Torbert, H.A., Gilliam, C.H., Fain, G.B., 2011. The importance of determining carbon sequestration and greenhouse gas mitigation potential in ornamental horticulture. HortScience 46, 240–244.

Martínez-Eixarch, M., Alcaraz, C., Viñas, M., Noguerol, J., Aranda, X., et al., 2018. Neglecting the fallow season can significantly underestimate annual methane emissions in Mediterranean rice fields. PLOS ONE 13(8), e0202159.

McSwiney, C.P., Robertson, G.H., 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (*Zea mays* L.) cropping system. Global Change Biol. 11, 1712-1719.

Minamikawa, K., Sakai, N., Yagi, K., 2006. Methane emission from paddy fields and its mitigation options on a field scale. Microbes Environ. 21(3), 135–147.

Moslehuddin, A.Z.M., Hussain, M.S., Saheed, S.M., Egashira, K., 1999. Clay mineral distribution in correspondence with agroecological regions of Bangladesh soils. Clay Sci. 11, 83–94

Naser, H.M., 2005. Evaluating the Status of Greenhouse Gas Budgets of Paddy Fields in Central Hokkaido, Japan (Ph.D. thesis). Hokkaido University, Sapporo, Japan, p. 87.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C., 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Europ. J. Agronomy 65, 40–51.

Neue, H., 1993. Methane emission from rice fields: Wetland rice fields may make a major contribution to global warming. BioScience 43 (7), 466-73.

Pathak, H., Li, C., Wassmann, R., 2005. Greenhouse gas emissions from Indian rice fields: calibration and upscaling using the DNDC model, Biogeosciences 2, 113–123

Patil, M.D., Das, B.S., 2013. Assessing the effect of puddling on preferential flow processes through under bund area of lowland rice field. Soil Till. Res. 134 61–71.

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., Halberg, N., 2013. An approach to include soil carbon changes in life cycle assessments. J. Clean. Prod. 52, 217-224.

Powlson, D.S., Stirling, C.M., Thierfelder, C., White, R.P., Jat, M.L., 2016. Does conservation agriculture deliver climate change mitigation through soil carbon sequestration in tropical agro-ecosystems? Agric. Ecosyst. Environ. 220, 164–174.

RMIT, 2007. Australian LCA Database–2007. Centre for Design, Royal Melbourne Institute of Technology, Melbourne, Australia.

Robertson, G.P., Grace, P.R., 2004. Greenhouse gas fluxes in tropical and temperate agriculture: the need for a full-cost accounting of global warming potentials. Environ. Dev. Sustain. 6, 51–63.

Ruser, R., Flessa, H., Schilling, R., Beese, F., Munch, J.C., 2001. Effect of crop-specific field management and N fertilization on N_2O emissions from a fine-loamy soil. Nutr. Cycling Agroecosys. 59, 117-191.

Ryu, J.H., Lee, J.S., Kim, K.H., Kim, G.Y., Choi, E.J., 2013. A case study to estimate the greenhouse–gas mitigation potential on conventional rice production system. Korean J. Soil Sci. Fert. 46, 502–509

Sander, B.O., Samson, M., Pearl, B., Sanchez, K.P., Valencia, E.A.M., Demafelix, Buresh, R.J., 2018. Contribution of fallow periods between rice crops to seasonal GHG emissions: effect of water and tillage management, J. Soil Sci. Plant Nutr. 64, 200-209.

Sass, R.L., 2002. Spatial and temporal variability in methane emissions from rice paddies: Implications for assessing regional methane budgets. Nutr. Cycling Agroecosyst. 64, 3-7.

Saunders, C., Barber, A., Taylor, G., 2006. Food miles – Comparative energy/emissions performance of New Zealand's Agriculture Industry. Research Report No. 285, July 2006.

Savant, N.K., De Datta, S.K., 1982. Nitrogen transformations in wetland rice soils. Adv. Agron. 35, 241-302.

So, H.B., Dalal, R.C., Chan, K.Y., Menzies, N.M., Freebairn, D.M., 2001. Potential of conservation tillage to reduce carbon dioxide emission in Australian soils. In: Mohtar, D.E., Steinhardt, G.C. (Eds.), Sustaining the Global Farm. Selected papers from the 10th International Soil Conservation Organization Meeting. Pp. 821– 826.

Steinbach, H.S., Alvarez, R., 2006. Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in Pampean agroecosystems. J. Environ. Qual. 35, 3–13.

Suh, S., 2004. Material and energy flows in industry and ecosystem network. Centre for Environmental Science, University of Leiden, The Netherlands.

Thanawong, K., Perret, S.R., Basset-Mens, C., 2014. Eco-efficiency of paddy rice production in Northeastern Thailand: a comparison of rain-fed and irrigated cropping systems. J. Clean. Prod. 73, 204–217.

Towprayoon, S., Smakgahn, K., Poonkaew, S., 2005. Mitigation of methane and nitrous oxide emissions from drained irrigated rice fields. Chemosphere 59, 1547-1556.

UN-FCCC, 2017. UN Climate Change annual report 2017, (available at https://unfccc.int/resource/annualreport/media/UN-Climate-AR17.pdf). (Accessed on 22 July 2018).

Ussiri, D.A.N., Lal, R., Jarecki, M.K., 2009. Nitrous oxide and methane emissions from long-term tillage under a continuous corn cropping system in Ohio. Soil Till. Res. 104, 247-255.

Vanden Bygaart, A.J., Angers, D.A., 2006. Towards accurate measurements of soil organic carbon stock change in agroecosystems. Can. J. Soil Sci. 86, 451–463.

Vetter, S.H., Sapkota, T.B., Hillier, J., et al., 2017. Greenhouse gas emissions from agricultural food production to supply Indian diets: implications for climate change mitigation. Agric. Ecosyst. Environ. 237, 234–241.

Wang, M., Xia, X., Zhang, Q., Liu, J., 2010. Life cycle assessment of a rice production system in Taihu region, China. Int. J. Sust. Dev. World 17, 157–161.

WB, 2014. Price level ratio of PPP conversion factor (GDP) to market exchange rate. From http://data.worldbank.org/indicator/PA.NUS.PPPC.RF. (Accessed on 12.09.2014)

Weller, S., Kraus, D., Ayag, K.R.P., Wassmann, R., Alberto, C.R., Butterbach-Bahl, K., Kiese, R., 2014. Methane and nitrous oxide emissions from rice and maize production in diversified rice cropping systems. Nutr. Cycl. Agroecosyst. 101, 37–53.

Wells, C., 2001. Total Energy Indicators of Agricultural Sustainability: Dairy Farming Case Study, Wellington: Ministry of Agriculture and Forestry, 2001.

WRI (World Resource Institute) and WBCSD, 2013.Technical Guidance for Calculating Scope 3 Emissions. Supplement to the Corporate Value Chain (Scope 3) Accounting & Reporting Standard. (Version 1.0), 2013. World Resources Institute & World Business Council for Sustainable Development, 2013. (Accessed on 12 October 2014).

Xu, X., Zhang, B., Liu, Y., Xue, Y., Di, Y., 2013. Carbon footprints of rice production in five typical rice districts in China. Acta Ecologica Sinica, 33, 227–232.

Yadav, G.S., Das, A., Lal, R., Babu, S., Meena, R.S., Saha, P., Singh, R., Datta, M., 2018. Energy budget and carbon footprint in a no-till and mulch based rice–mustard cropping system. J. Clean. Prod. 191, 144-157.

Yagi, K., Tsuruta, H. Kanda, K., Miami, K., 1991. Manual of CH_4 and N_2O flux measuring. Collected papers on environmental planning. 7, 143–158.

Zebarth, B.J., Rochette, P., Burton, D.L., 2008. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. Can. J. Soil Sci. 88, 197-205.

Zhang, D., Shen, J.B., Zhang, F.S., Li, Y.E., Zhang, W.F., 2017. Carbon footprint of grain production in China. Scientific Reports 7, 1–11.

Highlights

- We investigate the life cycle greenhouse gas emissions of a rice-based cropping system
- Minimum soil disturbance with current or increased residue retention reduced GHGs
- Irrigated or rainfed rice were dominant contributors to cropping system life cycle GHG
- On-farm CH_4 was the hotspot of GHG emissions