Environmental and economic assessment of paddy based cropping systems in Middle Indo-Gangetic plains, India

Rajeev Sinha, Peeyush Soni, Sylvain Roger Perret

To cite this version:

Rajeev Sinha, Peeyush Soni, Sylvain Roger Perret. Environmental and economic assessment of paddy based cropping systems in Middle Indo-Gangetic plains, India. Environmental and Sustainability Indicators, 2020, 8, pp.100067. 10.1016/j.indic.2020.100067 . hal-04669469

HAL Id: hal-04669469
https://hal.inrae.fr/hal-04669469v1

Submitted on 8 Aug 2024

Environmental and Sustainability Indicators 8 (2020) 100067

Contents lists available at ScienceDirect

Environmental and Sustainability Indicators

journal homepage: www.journals.elsevier.com/environmental-and-sustainability-indicators/

Environmental and economic assessment of paddy based cropping systems in Middle Indo-Gangetic plains, India

Rajeev Sinha [a], Peeyush Soni [b,*], Sylvain Roger Perret [c]

[a] Applications Technology, Corteva Agriscience, Zionsville, INpolis, Indiana, USA
[b] Agricultural and Food Engineering Department, Indian Institute of Technology Kharagpur, 721302, India
[c] CIRAD, Univ Montpellier MUSE, UMR G–EAU, F-34398, Montpellier, France

ARTICLE INFO

Keywords:
Paddy
Wheat
Potato
GHG emission
Eco–efficiency

ABSTRACT

The aim of this research was to analyze greenhouse gas (GHG) emission in paddy rice–based cropping systems of the Middle Indo–Gangetic plains in India. Two paddy rice–based systems, namely paddy rice–wheat (PW) and paddy rice–potato–fallow (PP), were studied for GHG emission, net return (NR) and eco–efficiencies (EE). The PW (3354.9 ± 133.7 kgCO$_2$eq ha^{-1} y^{-1}) and PP (5096.3 ± 11.6 kgCO$_2$eq ha^{-1} y^{-1}) systems were observed to have significantly different GHG emissions. The most significant contributor to the GHG pool was fertilizer production, contributing about 40.3% and 46.1% in the PW and PP systems, respectively. On–farm direct nitrous oxide (N_2O) emission was the second largest contributor to the GHG emission pool. Farm sizes significantly affected GHG emissions in the PW system, and medium farms ($>$3 ha) had higher GHG emission compared to other farm sizes. While the NR in the PP system was significantly higher (2504.3 ± 65.6 USD ha^{-1} y^{-1}) compared to the PW system (1687.1 ± 90.6 USD ha^{-1} y^{-1}) owing to better market prices of potato crop, it also performed slightly better on the EE scale (0.52 and 0.50 USD kgCO2 eq^{-1}, respectively). In the PW system, comparatively larger farms were more eco–efficient than the smaller farms. However, the EE values for different farm sizes in the PP system were numerically and statistically similar. The results reported in this study may be beneficial to farmers to make decisions pertinent to sustainable management of agro inputs in paddy–based cropping systems and collective farming. Moreover, these results can even be utilized by governmental and non–governmental organizations for deciding support/subsidies to farmers in the study area.

1. Introduction

With more and more people getting added to our planet, agricultural sector is under immense pressure to produce sufficient food with continuously reducing arable land (World Bank, 2008). To cope up with this, agriculture has changed its face considerably in the past decade and has shifted towards an enterprise which is more energy dependent and energy intensive. With the shift, there has been an increased concern about the environmental impacts caused by energy intensive farming (Astier et al., 2014). Energy intensive farming has contributed to increased greenhouse gas (GHG) emission and overall global warming. GHG emission from agricultural production systems have increased more than two–folds in the last 55 years (FAO STATS, 2019). GHG emissions from crops and livestock production systems witnessed about 14% increase from 2001 to 2011, and the developing countries were the main contributors to this increase (Seguin et al., 2007). Asian countries alone contributed about 44% of the total agriculture–related GHG emissions in 2011. Application of synthetic fertilizer and methane (CH_4) emission from submerged paddy fields accounted for 13% and 10% of the total agricultural emission, respectively (Kourous, 2014). The farmers in developing countries lack knowledge about excessive and injudicious use of agricultural inputs, which may cause environmental impacts in terms of GHG emission. Energy use and subsequent GHG emission are dependent on the type of crops grown and the cropping system followed. Thus, quantifying GHG emissions from different cropping systems may provide some insights regarding the environmental impacts created by such systems and would allow farmers to make pertinent management decisions, if warranted.

Paddy rice–based farming systems are quite common in the Indian subcontinent. A suitable topography, favorable climatic conditions, soil types and prevalence of monsoon bringing enough rains during the kharif season are required for paddy rice cultivation. India comes in the pool of

---

[*] Corresponding author.
E-mail address: soni@iitkgp.ac.in (P. Soni).

https://doi.org/10.1016/j.indic.2020.100067
Received 3 April 2020; Received in revised form 4 August 2020; Accepted 4 September 2020
Available online 8 September 2020

major paddy rice producing nations of the world and accounts for about 21% of the total white paddy rice production. However, paddy rice–based farming systems account for extensive use of direct as well as indirect forms of energy (Soni et al., 2018). The direct sources comprise of on–farm use of agricultural machinery and equipment. The indirect sources constitute the energy used in different processes like agrochemical and machinery production (Statistical Yearbook of India, 2012). Increasing the use of agricultural inputs has resulted in emission of GHGs like carbon dioxide (CO_2), nitrous oxide (N_2O) and CH_4 into the atmosphere. Use of fuel accounts for CO_2 emission and the use of nitrogen (N) fertilizers causes direct as well as indirect N_2O emissions. Moreover, flooded paddy rice fields emit high concentration of CH_4 due to anaerobic soil conditions (Yao et al., 2009). As per FAO reports, India ranks second (contributing about 21%) in the world for paddy rice–based CO_2 (equivalent) emission followed by China (FAO STAT, 2019).

Several studies have reported the GHG emission potential of different crops and pertinent management practices worldwide. In a recent study, Sarauskis et al. (2019) assessed the impact of different fertilization methods on GHG emission in wheat–barley–red clover system. A 10–fold increase in GHG emission was observed when farmyard manure was used as fertilizer compared to green manure. In another study, GHG emission from different rice tillage systems in China was evaluated (Ahmad et al., 2009). The study reported a higher global warming potential (GWP) for conventional tillage compared to no–tillage system. Similarly, about 20% higher GHG emission from castor grown on pigeon pea residue compared to pigeon pea grown on castor residue was reported by Pratibha et al. (2015). In a similar study conducted in Iran, about 9% reduction in GHG emission from potato production using efficient on-farm energy usage was reported by Khoshnevisan et al. (2013).

Field crop production in South Africa resulted in 5.2 million tonnes of CO_2eq emission in 2012. Application of synthetic fertilizers (57%) and lime (30%) and retaining crop residues (13%) in the field were the major contributors to the emission pool (Kggwane et al., 2016; FAOSTAT, 2019). A meta–analysis of global GHG data for cereal crops in 2011 reported a GWP of 3757, 662 and 1399 kgCO_2eq ha^{-1} $season^{-1}$ for rice, wheat, and maize, respectively (Linquist et al., 2011). There have been several studies evaluating GHG emission potential for different crops, however, there has been limited number of studies quantifying GHG emission from different crops in Indian conditions. Using InfoCrop simulation model, Bhatia et al. (2012) reported the GWP of about 2096.0 and 1588.3 kgCO_2eq ha^{-1} for rice and wheat production systems in India, respectively. Moreover, a very few studies (Bhatia et al., 2005) reported GWP for a cropping system in Indian conditions. To the best of our knowledge, none of the reported studies have compared GHG emission potential for different cropping systems.

To achieve the goal of sustainable intensified crop production, there is a need for adopting practices owing to improved yields, but at the same time has GHG mitigation as a must have option. This could only be achieved if there is a sufficient knowledge of the pertinent farming operations, different agro–inputs and the GWP associated with the systems. However, this seems to be a far–sighted goal as the primary concern of farmers is profit making notwithstanding the environmental impacts caused. This is especially relevant in crop production systems in developing countries. Therefore, taking into consideration only the GWP of cropping systems and agricultural practices will not provide a clear picture. GWP must be related to economics of production to arrive at some sort of trade–off. While this area has been very lightly explored, eco–efficiency (EE) is frequently used as an indicator which relates the environmental impact with economics of the production system. It is defined as the ratio of economic creation to ecological destruction (Sailing et al., 2002). While the economic creation is quantified in terms of economic return, the ecological destruction is quantified in terms of GHG emission.

Therefore, this study was carried out to study the environmental impact and economic performance of the two most followed paddy rice–based cropping systems of the Middle Indo–Gangetic plain in India. The specific objectives of the study were to evaluate the a) GHG emission potential of a paddy rice–wheat (PW) and paddy rice–potato–fallow (PP) cropping systems in the region, b) EE of the two system and c) effect of farm size classification on the GHG emission, NR and EE.

2. Materials and methods

2.1. Site description

The selected sites for the study were Patna and Allahabad, the two important cities in the Middle Indo–Gangetic plains in India. Allahabad (24°47′ N, 81°19′ E) and Patna (25°36′N, 85°7′E) are drained by river Ganges and have similar alluvial soil profile. Both sites have a humid sub–tropical climate with a hot summer from the end of March to early June. South–eastern monsoons prevail from the end of June to early October, and winter lasts from the middle of November to February. Both areas are characterized by three seasons namely hot dry summer, cool dry winter, and warm humid monsoon. In both areas, farmers practice rainfed as well as irrigated farming, depending on the monsoon season.

Paddy rice is grown in *Kharif* (monsoon) season, while wheat and potato are grown in *Rabi* (dry) season in both the areas under study. Most of the pre–harvest processes (tillage, seeding, weed management and irrigation) in the study areas are mechanized and inorganic fertilizers are preferred over the organic ones. Manual method of harvesting for paddy rice and wheat is usually employed as the farm sizes are not economically suitable for large harvesting machines, with a very few farmers owning/custom hiring combine harvesters. Custom hiring of farm machineries is most common in the study sites and small farmers also have access to farm mechanization through custom hiring. However, small farms are mostly household farms and household labourer is the prime source of agricultural labour. Nearly all the farmers have pumping sets to pump groundwater for irrigation in case of delayed monsoon. Overall, the farmers in the study areas have adopted energy intensive farming for obtaining a higher profit from their production system.

2.2. Data collection

Data were collected through a direct face to face interview with the farming community engaged in the PW and PP systems. Primary as well as secondary data were collected for this study. Primary data was collected using a set questionnaire, included those factors that emitted GHGs and affected the economic performance. Primary data collected were related to the amount of agricultural inputs like fertilizer, pesticide and herbicides, flooding period of paddy rice fields, amount of biomass burning, fuel consumption, number of cattle, and manure management practices. The secondary data were collected from various published literature. In addition, Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 2006) and 2019 Refinements to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2019) were followed to estimate GHG emission. A total of 53 farmers engaged in the PW and PP cropping systems were interviewed in the selected areas.

2.3. GHG emissions calculation

2.3.1. Emission from production and combustion of fossil fuel and electricity use

The amount of GHG emissions from the use of fossil fuel and electricity use was calculated using Eqs. (1) and (2).

$$\text{Emissions from fossil fuel} = \Sigma\ \{FC \times (EF_1 + EF_2 + EF_3)\} \tag{1}$$

$$\text{Emissions from electricity use} = \Sigma\ (TCUE \times EF_{electricity}) \tag{2}$$

where, FC = total fuel consumption (m^3), EF_1 = CO_2 emission factor (kgCO_2 m^{-3} fuel consumed), EF_2 = CH_4 emission factor (kgCH_4 m^{-3} fuel consumed), EF_3 = N_2O emission factor (kgN_2O m^{-3} fuel consumed), TCUE = total commercial unit of electricity consumed (kWh) and

Table 1
Emission factors for calculation of GHG emission due to fossil fuel use.

S. No.	Fuel type	$kgCO_2\ m^{-3}$ (EF_1)	$kgCH_4\ m^{-3}$ (EF_2)	$kgN_2O\ m^{-3}$ (EF_3)
1.	Diesel	2697.19	0.38	0.07
2.	Petrol	2319.43	0.33	0.06

$EF_{electricity}$ = emission factor for electricity (kg $[kWh]^{-1}$).

Table 1 provides the emission factor values for calculation of GHG emission (EPA, 2014). Using the data for total fuel consumption for the individual cropping systems, and the various emission factors, the net GHG emission from fossil fuel use was calculated. The country specific emission factor for electricity consumption (0.7429 $kgCO_2eq\ [kWh]^{-1}$) was taken from previously reported literature (Carbon Footprint, 2019).

2.3.2. Emission from production, packaging, storage, and transport of agrochemicals

The proportion of basic nutrients (like N, P, K, S and Zn) in fertilizers used was calculated using their chemical composition and molecular formula. For the chemicals, as suggested, each chemical element was multiplied by a conversion factor (0.50 for herbicide, and 0.25 for insecticides and plant growth promoters) for calculating the associated GHG emission (Rab et al., 2008). Lal (2004) presented emission factors for calculating CO_2 emissions from the production, packaging, storage and transportation of agrochemical. These emission factors were multiplied by the actual nutrient or active ingredient (a.i.) concentration to get the corresponding CO_2 emissions. The emission factors for various nutients and agrochemicals which were further modified by Maraseni and Cockfield (2011) for converting the C–equivalents to CO_2 equivalents is provided in Table 2.

2.3.3. CH_4 emission from submerged paddy rice fields

In flooded paddy rice fields anaerobic decomposition of organic material produces CH_4 which escapes mainly in the form of air bubbles to the atmosphere and by transport through plants. The amount of CH_4 emission from the paddy rice fields depends on crop species, number and duration of harvests, soil type and temperature, irrigation method and fertilizer use (IPCC, 2019). CH_4 emission from submerged paddy rice fields was calculated using Tier 1 equation of IPCC (Eq. (3)) as given below.

$$(\mathrm{CH4})\mathrm{Paddy}-\mathrm{rice} = \sum_{i,j,k}\left(EF_{i,j,k}\times t_{i,j,k}\times A_{i,j,k}\times 10^{-6}\right) \tag{3}$$

where, $(CH_4)_{Paddy\text{-}rice}$ = total CH_4 emissions from paddy rice cultivation ($GgCH_4\ y^{-1}$), $EF_{i,j,k}$ = a daily emission factor for i, j and k conditions, $kgCH_4\ ha^{-1}\ day^{-1}$, $t_{i,j,k}$ = total period of submergence, day, $A_{i,j,k}$ = annual harvested area of rice for i, j and k conditions, ha y^{-1}, and i, j and k represent different ecosystems, water regime, type and amount of organic amendments, and other conditions under which CH_4 emissions from rice may vary.

The adjusted daily emission factor (EF_i) is calculated using Tier 1 methodology as provided in equation (4) below.

$$EF_i = EF_c \times SF_w \times SF_p \times SF_o \tag{4}$$

Table 2
Emission factors for various nutrients and agrochemicals.

Fertilizer	$kgCO_2\ kg^{-1}$ f.e.	Chemical	$kgCO_2\ kg^{-1}$ a.i.
N	4.77	Insecticide	18.70
P	0.73	Herbicide	23.10
K	0.55	Pesticide	14.30
S	0.37		
Zn	0.37		

*f.e. = fertilizer element; a.i. = active ingredient.

where, EF_i = adjusted daily emission factor for a particular harvested area, EF_c = baseline emission factor for continuously flooded fields without organic amendments (0.85 for South Asia), SF_w = scaling factor to account for the differences in water regime during the cultivation period (0.55 for irrigated water regime with multiple drainage periods), SF_p = scaling factor to account for the differences in water regime in the pre-season before the cultivation period (0.89 for a >180 days non-flooded pre-season) and SF_o = scaling factor should vary for both type and amount of organic amendment applied (IPCC, 2019).

2.3.4. N_2O emission due to nitrogen fertilizer application in soil

Oxides of N accounts for around 6% of the total observed global warming. About 80% of the oxides of N are produced by agricultural activities, and around two–third of this is emitted from agricultural soils (Dalal et al., 2003). The IPCC has set a default emission factor of 1.25% N_2O–N emissions per kg of applied N, which varies from the experimentally calculated values in different regions of the world. The IPCC provides methodologies for estimating national inventories of GHG emissions due to anthropogenic activities. The methodologies include guidelines for GHG inventory arrangements and management, data gathering, compilation and reporting. The methods are provided in three tiers which are classified based on the level of complexity, with Tier–1 as the simplest and Tier–3 as the most complex. The IPCC provides guidelines for the calculations of both direct and indirect emissions using any of the three tiers, and the selection of a specific tier depends on data availability. In this study, the direct as well as the indirect emissions were calculated using the country specific emission factors. The indirect N_2O emissions from soil was calculated by considering only the use of synthetic fertilizers and application of organic manure to the soil. The formula for this calculation is taken from IPCC 2019 guidelines which remains unchanged from the 2006 guidelines. N_2O from atmospheric decomposition from managed soil due to N–volatilization and N_2O–N due to leaching and runoff is given by equations (5) and (6).

$$N_2O_{(ATD)} - N = \left[(F_{SN} \times Frac_{GASF}) + ((F_{ON} + F_{PRP}) \times Frac_{GASM})\right] \times EF_4 \tag{5}$$

$$N_2O_{(L)} - N = (F_{SN} + F_{ON} + F_{PRP} + F_{CR} + F_{SOM}) \times Frac_{LEACH-(H)} \times EF_5 \tag{6}$$

where, $N_2O_{(ATD)}$–N and $N_2O_{(L)}$–N are the annual amount of N_2O–N produced from atmospheric decomposition and leaching, respectively in $kgN_2O\text{–}N\ y^{-1}$. F_{SN}, F_{ON}, F_{PRP}, F_{CR} and F_{SOM} represent the amount of N applied or incorporated to the soil by the use of synthetic fertilizer, organic sources, urine and dung deposition, crop residue and mineralization of soil organic matter, respectively, in kgN y^{-1}. $Frac_{GASF}$ and $Frac_{GSM}$ are fractions of synthetic fertilizer N and organic N, respectively, that volatizes as ammonia (NH_3) and NO_x in kgN–volatilized $(kgN)^{-1}$ applied. $Frac_{LEACH}$ is the faction of all the N added to/mineralized in managed soils in regions where leaching or runoff occurs in kg N $(kgN)^{-1}$ addition. EF_4 and EF_5 are the emission factors for N_2O emissions from atmospheric decomposition on soil and water surfaces, and from leaching and runoff, respectively.

Conversion of N_2O–N emissions to N_2O emissions for reporting purpose is performed by using equation (7). The indirect emissions due to atmospheric decomposition and leaching are calculated using the IPCC Tier–2 methodology, using the country specific emission factors. The country specific emission factors for the calculations are taken from Bhatia et al. (2005) (Table 3).

$$N_2O_{ATD/LEACH} = N_2O_{ATD/LEACH} - N \times {}^{44}\!/\!{}_{28} \tag{7}$$

2.3.5. Emissions due to the production and use of farm machinery

The estimate of GHG emission from manufacturing of farm machineries was based on machinery weight (GHG kg^{-1} machine weight). In addition, Maraseni et al. (2007) estimated GHG emission from machinery use to be 14.4% of the emissions due to fossil fuel use. Thus, the GHG emissions from machinery production and use for on–farm operations

Table 3
Country specific emission factors for India.

Parameter	IPCC coefficients*	Revised coefficients
EF_c, seasonally integrated emission factor for continuously flooded fields	0.85 $kgCH_4$ ha^{-1} d^{-1}	State–specific coefficients
SF_w (scaling factor for different water ecosystems)		
Irrigated: Continuous flooding	1.0	1.0
Irrigated: Single drainage period	0.71	0.6
Irrigated: Multiple drainage period	0.55	0
Rainfed: Deepwater	0.54	0.8
Rainfed: Drought prone	0.16	
Rainfed: Deep water	0.06	
EF_1 (N_2O emission from applied fertilizer, %)	1.25	0.7
EF_2 (N_2O emission from organic soil, %)	16	16
EF_4 (N_2O emission from volatilized N from fertilizer and manure, %)	1.1	0.5
EF_5 (N_2O emission from leached and run–off N from fertilizer and manure, %)	2.5	0.5
$Frac_{GASF}$ (gas loss through volatilization from inorganic fertilizer, %)	11	15
$Frac_{GASF-AM}$ (gas loss through volatilization from manure; %)	21	15
$Frac_{leach}$ (leaching loss of N from applied fertilizer and manure, %)	24	10

*Default coefficients based on 2019 refinements to the 2006 IPCC guidelines for national GHG inventories.

were calculated by multiplying the emission due to fossil fuel and its appropriate weight factor.

2.4. *Net return*

Net return is defined as the total economic gain to a farmer following a production system. This is an important indicator of a farmer's economic perspective (Eq. (8)).

$$\text{Net Return (USD ha}^{-1}\text{ y}^{-1}) = \text{Gross Income (USD ha}^{-1}\text{ y}^{-1}) \text{ -Total Annual Input Cost (USD ha}^{-1}\text{ y}^{-1}) \quad (8)$$

Where, gross income is calculated by multiplying the total crop produced by its unit price, and total annual input cost represents all the fixed as well as the variable costs incurred during crop production.

2.5. *Eco–efficiency*

EE is an approach focused at pooling resource use and pollutant release from a given economic activity (Eq. (9)). It is defined as the ratio of economic return from an activity to the environmental degradation or the impact it is creating. The aim of sustainable agriculture is to increase its EE by lowering the environmental impacts like energy use and GHG emissions of agriculture while increasing the economic output (Muller and Sturm, 2001; Gomez−Limon et al., 2012).

$$\text{Eco–efficiency (USD kgCO}_2\text{eq}^{-1}) = \text{Net Return (USD ha}^{-1}\text{)/GHG emission (kgCO}_2\text{eq ha}^{-1}) \quad (9)$$

2.6. *Farm size classification*

All the indicators (GHG emission, net return and eco–efficiency) were calculated and analyzed for the two cropping systems under different farm sizes. The farms were categorized as marginal (<1 ha), small (1–3 ha) and medium (>3 ha).

2.7. *Data analysis*

All the response variables (i.e., GHG emission, NR and EE) were compared for the two systems using two sample t–test. The data pertinent to GHG emission, net return and eco-efficiency were analyzed using analysis of variance (ANOVA) followed by post–hoc multiple comparison of means using Tukey's Honest Significant Difference (HSD) test at a significance level of $\alpha = 0.05$ (Sinha et al., 2019). All the statistical analyses were carried out in R (ver. 3.4.0, R Foundation for Statistical Computing, Vienna, Austria.

3. Results and discussion

3.1. *GHG emissions*

GHG emission in the PP system (5096.3 ± 111.6 $kgCO_2eq\ ha^{-1}\ y^{-1}$) (mean ± standard error) was significantly higher compared to the PW system (3354.9 ± 133.7 $kgCO_2eq\ ha^{-1}\ y^{-1}$) (*Two sample t–test,* $t_{95} = 9.99$, $p < 0.001$). Various agricultural practices that contributed to the GHG pool and their % contribution to the total GHG emission in the two systems have been reported in Table 4. For paddy rice, wheat and potato crops, prior studies have reported individual GHG emissions (Ahmad et al., 2009; Koga and Tajima, 2011; Pishgar–Komleh et al., 2012; Khoshnevisan et al., 2013a, b, c; Ma et al., 2013; Pandey et al., 2013; Soltani et al., 2013; Soni et al., 2013; Xue et al., 2018). Most of these studies reported GHG emission to be in line with the results reported in this study. Also, there were a few studies thatreported a slightly lower value of GHG emission. In Iran, a total GHG emission of about 0.99 $tonCO_2eq\ ha^{-1}$ was reported in potato production (Komleh et al., 2012) which was lower compared to what reported in the current study. This was possibly due to non–inclusion of direct and indirect N_2O emissions and a lower rate of fertilizer application in potato production. In a similar study, the total GHG emission from potato production in Iran was observed to be 2.2 $tonCO_2eq\ ha^{-1}$ with electricity as the main contributor followed by fertilizer use (Khoshnevisan et al., 2013b). Similarly, a total emission of 3.5 $tonCO_2eq\ ha^{-1}$ from paddy rice fields was reported by Koga and Tajima (2011) when the straw was removed from the field. For wheat production in Australia, a total emission of about 0.6 $tonCO_2eq\ ha^{-1}$ under conventional tillage as compared to 0.5 $tonCO_2eq\ ha^{-1}$ in reduced tillage condition was reported by Maraseni and Cockfield (2011).

Fertilizer production was the main contributor to the GHG pool in the PP and PW systems, contributing around 46.1% and 40.3%, respectively. It was observed that there was a significantly higher GHG emission due to fertilizer use in the PP system (2348.6 ± 85.7 $kgCO_2eq\ ha^{-1}\ y^{-1}$) compared to the PW system (1352.7 ± 101.8 $kgCO_2eq\ ha^{-1}\ y^{-1}$) (*Two-way ANOVA,* $F_{7,776} = 34.57$, $p < 0.001$). This difference in the GHG emission may have resulted from a high use of N–fertilizers in the PP system. The farmers in the study area mainly used urea (46% N) and DAP (18% N). Similar trend was reported for the use of P–based fertilizers.

Table 4
Overall GHG emission ($kgCO_2eq\ ha^{-1}\ y^{-1}$) (mean ± standard error) of different inputs in the two systems.

Input	PP		PW	
	GHG emission	%	GHG emission	%
Fuel use	489.5 ± 32.1^{ef}	9.6	374.1 ± 27.3^{fg}	11.1
Electricity	209.2 ± 34.5^{ghi}	4.1	51.4 ± 11.9^{ij}	1.5
Machinery use	70.5 ± 4.6^{ij}	1.4	53.9 ± 3.9^{ij}	1.6
Fertilizer production	2348.6 ± 85.7^{a}	46.1	1352.7 ± 101.8^{b}	40.3
Agrochemical production	233.4 ± 52.4^{ghi}	4.6	19.1 ± 5.6^{j}	0.6
Methane emission	623.9 ± 0.0^{de}	12.2	624.1 ± 0.0^{de}	18.6
Direct N_2O emissions	818.4 ± 0.0^{c}	16.1	691.7 ± 0.0^{cd}	20.6
Indirect N_2O emissions	302.7 ± 4.5^{fgh}	5.0	187.9 ± 8.9^{hij}	5.6

Different letters associated with mean values in rows show significant difference at $\alpha = 0.05$.

The use of fertilizer was observed to be dependent on the financial condition of the farmers. A farmer with strong financial background used more fertilizers on his farm. Similar studies on GHG emission assessment (Ahmad et al., 2009; Koga and Tajima, 2011; Khoshnevisan et al., 2013a; Zhang et al., 2016; Xue et al., 2018) also reported that fertilizer manufacturing and production was the major contributor to the GHG pool in respective crop production systems.

GHG emission from the production of agrochemicals was also reported to be significantly higher in the PP system (233.4 ± 52.4 $kgCO_2eq$ ha^{-1} y^{-1}) compared to the PW system (19.1 ± 5.6 $kgCO_2eq$ ha^{-1} y^{-1}) ($p < 0.001$) (Table 4). Potato, being a specialty crop, was treated with a higher quantity of pesticides as it is highly susceptible to various fungal and bacterial pathogens. However, for paddy rice and wheat crops, it was observed that only progressive farmers in the study area used agrochemicals. Higher amount of N– and P–fertilizers also contributed towards a numerically higher direct emissions of nitrous oxide (N_2O) in the PP system (818.4 ± 0.0 $kgCO_2eq$ ha^{-1} y^{-1}) compared to the PW (691.7 ± 0.0 $kgCO_2eq$ ha^{-1} y^{-1}) system (Table 4). A similar trend was reported with the PP (302.7 ± 4.5 $kgCO_2eq$ ha^{-1} y^{-1}) system having a numerically higher indirect N_2O emission compared to the PW (187.9 ± 8.9 $kgCO_2eq$ ha^{-1} y^{-1}) system. However, it is worth mentioning that the studies reporting GHG emission for either a crop or a cropping system (Pathak et al., 2002; Ahmad et al., 2009; Zhang et al., 2010; Koga and Tajima, 2011; Khoshnevisan et al., 2013a) did not calculate the on–farm direct and indirect emissions using IPCC methodology.

Fuel and electricity use was the next major contributor to the GHG emissions in the two systems which contributed about 9.6% and 11.1%, respectively, in the PP and PW systems (Table 4). In the PP system, a large fraction of farmers utilized manual method of potato harvesting which justifies a lower value of GHG emission due to the use of fuel and electricity in the PP system compared to the PW system. In addition, a large number of farmers utilized combine harvesters for harvesting wheat which also contribute to the GHG pool. While a large number of farmers in the PP system used potato digger–cum–elevator for the harvesting of potatoes, manual labor was used to pick potatoes from the farm. In most of the surveyed farms in both the systems, the farmers would do custom hiring of the machinery (e.g., potato digger, combine harvester etc.) or would seek help from a local university with such facility. Emission of CH_4 from submerged paddy rice fields was reported to be similar in the two systems which was probably because of similar water regimes and submergence period of paddy rice in the two systems. In most cases, the farmers would use an electric/fuel operated irrigation pumps to irrigate paddy and potato crops, which may justify a numerically higher use of fuel and electricity in the PP system. However, no significant differences in the mean values of these indicators were reported by the study (Table 4).

3.2. Effect of farm size on GHG emission

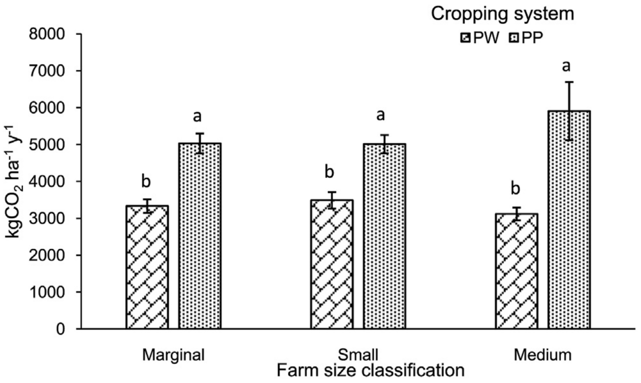

Fig. 1. GHG emission ($kgCO_2eq$ ha^{-1} y^{-1}) for paddy rice–wheat (PW) and paddy rice–potato–fallow (PP) systems in the two cropping systems (different lowercase letters with mean represent significant difference between the mean values; error bars represent standard error of mean).

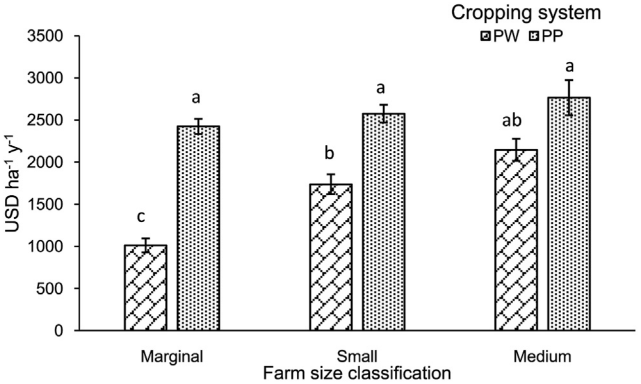

Fig. 2. Net return (USD ha^{-1} y^{-1}) for paddy rice–wheat (PW) and paddy rice–potato–fallow (PP) systems in the two cropping systems (different lowercase letters with mean represent significant difference between the mean values; error bars represent standard error of mean).

The effect of farm sizes on GHG emission in both the systems (i.e. PP and PW) was assessed for marginal, small and medium farm categories (Fig. 1). There were no significant differences observed in GHG emission pertinent to the PP system among marginal (5030.9 ± 269.9 $kgCO_2eq$ ha^{-1} y^{-1}), small (5009.5 ± 247.8 $kgCO_2eq$ ha^{-1} y^{-1}) and medium farms (5905.6 ± 788.9 $kgCO_2eq$ ha^{-1} y^{-1}) (*Two-way ANOVA,* $F_{2,93} = 2.75, p = 0.06$) (Fig. 2). However, the relatively larger farms (i.e., medium farms) had numerically higher GHG emission compared to relatively smaller farms (i.e., marginal and small farms). This difference was mainly because of a significantly higher amount of GHG emission from the fertilizer production category in the medium farms (Table 5). The results further indicated that GHG emission under abovementioned farm sizes in the PW system were numerically similar among marginal (3332.9 ± 184.6 $kgCO_2eq$ ha^{-1} y^{-1}), small (3487.9 ± 222.0 $kgCO_2eq$ ha^{-1} y^{-1}) and medium farms (3118.9 ± 170.9 $kgCO_2eq$ ha^{-1} y^{-1}) ($p > 0.05$) (Fig. 1). The small farm category in the PW system had the highest GHG emission (1433.9 ± 140.0 $kgCO_2eqha^{-1}y^{-1}$) and this was mainly due to an increased use of fertilizer reported by the farmers owning small farms (Table 6).

The results indicate that in the PP system the larger farms had higher GHG emissions, however, no significant differences were observed. This may be attributed to an increased use of farm inputs in larger farms. This can further be related to a better financial condition of the farmers owning comparatively larger farms. In the study area for the PP system,

Table 5
Overall GHG emission ($kgCO_2eq$ ha^{-1} y^{-1}) from different inputs under different farm size classification in the PP system.

Inputs	GHG emission (mean ± standard error)		
	Farm size classification		
	Marginal	Small	Medium
Fuel	494.2 ± 43.3^{def}	445.37 ± 51.9^{def}	645.6 ± 111.4^{cde}
Machinery use	71.2 ± 6.2^{g}	64.1 ± 7.5^{g}	93.0 ± 16.0^{g}
Electricity	134.2 ± 40.8^{fg}	355.2 ± 55.5^{defg}	94.7 ± 94.7^{g}
Agrochemical production	343.1 ± 86.9^{defg}	105.8 ± 22.7^{g}	36.1 ± 15.1^{g}
Fertilizer production	2245.4 ± 81.6^{b}	2309.57 ± 103.8^{b}	3462.6 ± 784.7^{a}
Methane emission	623.9 ± 0.0^{cdef}	623.9 ± 0.0^{cdef}	623.86 ± 0.0^{cde}
Direct N_2O emission	818.4 ± 0.0^{c}	818.4 ± 0.0^{cd}	818.4 ± 0.0^{cd}
Indirect N_2O emission	300.5 ± 11.0^{efg}	287.2 ± 6.5^{efg}	383.0 ± 70.3^{defg}

Different letters associated with mean values represent significant difference at α = 0.05.

Table 6
Overall GHG emission (kgCO$_2$eq ha^{-1} y^{-1}) from different inputs under different farm size classification in the PW system.

Input	GHG emission (mean ± standard error)		
	Farm size classification		
	Marginal	Small	Medium
Fuel	379.3 ± 36.5^{bcd}	415.0 ± 37.8^{bc}	259.8 ± 23.0^{bcde}
Electricity	72.9 ± 23.3^{cde}	43.6 ± 9.8d^{e}	46.4 ± 17.1^{de}
Machinery use	54.6 ± 5.3^{de}	59.7 ± 5.4^{de}	37.4 ± 3.3^{de}
Agrochemical production	25.0 ± 15.2^{e}	23.4 ± 6.6^{e}	3.9 ± 1.6^{e}
Fertilizer production	1294.0 ± 85.2^{a}	1433.9 ± 140.0^{a}	1292.2 ± 106.5^{a}
Methane emission	624.1 ± 0.0^{b}	624.1 ± 0.0^{b}	624.1 ± 0.0^{b}
Direct N$_2$O emission	691.7 ± 0.0^{b}	691.7 ± 0.0^{b}	691.7 ± 0.0^{b}
Indirect N$_2$O emission	191.1 ± 19.1^{cde}	196.3 ± 22.5^{cde}	163.3 ± 19.3^{cde}

Different letters associated with mean values represent significant difference at $\alpha = 0.05$.

scarcity of agricultural laborers was observed due to various government employment guarantee scheme like MNREGA (Mahatma Gandhi National Rural Employment Guarantee Act) resulting in an increased dependency on mechanized means which in turn resulted in higher GHG emission due to on–farm fossil fuel use. While labor shortage forced farmers with larger farms to adopt mechanized means, mechanizing the production process was not economically viable for farmers with comparatively smaller farms. Similar results were reported by Khoshnevisan et al. (2013a) where GHG emission in wheat production in Esfahan province of Iran was assessed using artificial neural networks. The study reported a higher emission in larger farms due to on farm fossil fuel use.

GHG emissions in the PP system was significantly higher than in the PW system in marginal (5030.9 ± 269.9 and 3332.9 ± 184.6 kgCO$_2$eqha^{-1}y^{-1}, respectively), small (5009.5 ± 247.8 and 3487.9 ± 222.0 kgCO$_2$eqha^{-1}y^{-1}, respectively) and medium (5905.6 ± 269.9 and 3118.9 ± 170.9 kgCO$_2$eqha^{-1}y^{-1}, respectively) farm categories ($p < 0.001$) (Fig. 1). This difference was mainly due to a significantly higher use of synthetic fertilizers and agrochemicals in the PP system, compared to the PW system. In addition, the farmers in the PP system had some support from a nearby university to custom hiring of agricultural machinery used in harvesting of paddy rice and potato. This may have been the reason for a higher GHG emission in fuel and machinery use (Tables 5 and 6).

3.3. Net return and the effect of farm size

This study reported that net return in the PP system (2504.3 ± 65.6 USD ha^{-1} y^{-1}) was significantly higher compared to the PW system (1687.1 ± 90.6 USD ha^{-1} y^{-1}) ($t_{89.8} = 8.44$, $p < 0.001$). The effect of farm sizes on net return was also assessed in both the paddy based systems. The study reported that in the PP system, the net return was statistically similar among the marginal (2422.1 ± 89.2 USD ha^{-1} y^{-1}), small (2573.9 ± 106.1 USD ha^{-1} y^{-1}) and medium farms (2764.0 ± 208.7 USD ha^{-1} y^{-1}) ($p > 0.05$) (Fig. 2). The results further indicated that for the PW system the net return in the small (1737.7 ± 118.3 USD ha^{-1} y^{-1}) and medium farms (2147.0 ± 130.9 USD ha^{-1} y^{-1}) were significantly higher compared to the marginal farms (1011.0 ± 83.1 USD ha^{-1} y^{-1}) ($p < 0.001$) (Fig. 2).

The farmers with comparatively larger land holdings (i.e., small and medium) were reported to have a higher net return in both the systems. Similar results were observed for potato production system in India where the return was reported to be 859.96 USD ha^{-1} (Kushwah and Singh, 2011). The net return from paddy was reported to be 600 USD ha^{-1}, and for wheat production it was reported to be around 340 USD ha^{-1} (Kushwah and Singh, 2011). It is worth mentioning that the increased use of machinery may have reduced the labor cost significantly and thus, there was an increase in the net return. These results may indicate that smaller land holdings are not economical in the study area. Therefore, a direct intervention/support package (for agro–inputs) is required on the part of the government and other non–governmental organizations directed towards farmers of relatively smaller farms. In addition, a community/collective farming approach could be beneficial to these farmers keeping in mind the higher returns fetched by the comparatively larger farms.

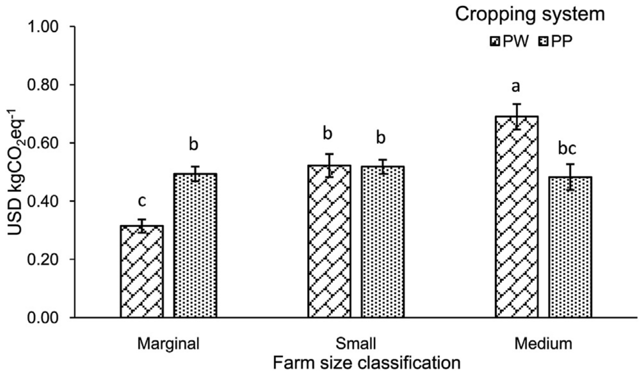

Fig. 3. Eco–efficiency (USD kgCO$_2$eq^{-1}) for paddy rice–wheat (PW) and paddy rice–potato–fallow (PP) systems in the two cropping systems (different lowercase letters with mean represent significant difference between the mean values; error bars represent standard error of mean).

3.4. Eco–efficiency and the effect of farm size

The eco–efficiency (EE) values in the PP (0.50 ± 0.02 USD kgCO$_2$eq^{-1}) and PW (0.52 ± 0.03 USD kgCO$_2$eq^{-1}) systems were statistically similar ($t_{77.7} = 0.63$, $p = 0.53$). The study further reported that the EE in the PW system was significantly different among the marginal (0.31 ± 0.02 USD kgCO$_2$eq^{-1}), small (0.52 ± 0.04 USD kgCO$_2$eq^{-1}) and medium farms (0.69 ± 0.04 USD kgCO$_2$eq^{-1}) ($p < 0.001$) (Fig. 3). These differences were significant mostly due to significant differences observed in total GHG emission and net return among different farm sizes (Figs. 1 and 2). However, there was no evidence that suggested any significant difference in EE value in the PP system among the different farm categories (Fig. 3). Further, the results indicated that the EE of marginal farms in the PP system (0.49 ± 0.02 USD kgCO$_2$eq^{-1}) was significantly higher compared to the PW system (0.31 ± 0.02 USD kgCO$_2$eq^{-1}) ($p < 0.001$). In the small farm category, the PW (0.52 ± 0.04 USD kgCO$_2$eq^{-1}) system had a numerically similar EE value compared to the PP system (0.52 ± 0.02 USD kgCO$_2$eq^{-1}). However, a significantly higher EE value was observed in the PW (0.69 ± 0.04 USD kgCO$_2$eq^{-1}) system compared to the PP (0.48 ± 0.04 USD kgCO$_2$eq^{-1}) system under medium farm category (Fig. 3). These results indicate that comparatively larger farms are more eco–efficient for paddy-based cropping systems in the study area compared to the smaller farms. In contrast, farm size did not had any significant effect of EE under PP system.

3.5. Discussion

The GHG emission in the PP system was found to be about 51.9% higher than that of the PW system. A significantly higher emission in the PP system compared to the PW system was mainly because of a resource intensive production in the former system. While the PP system had a significantly higher GHG emission, the emission pertinent to fertilizer use (i.e., production, direct and indirect N2O emissions) contributed about 67.2% and 66.5% in the PP and PW systems, respectively. This indicates that there may be some scope to reduce the carbon footprint of the respective production systems with a judicious use of fertilizer in the crop production systems. During the initial questionnaire, a majority of farmers communicated that they believed a higher fertilizer input would result in a greater yield and were not aware of the recommended rate and timing of fertilizer application. Therefore, creating awareness among the farmers about proper fertilizer application would definitely reduce the environmental impact created by such systems. In addition, a positive intervention from various governmental and non-governmental agencies is recommended to support farmers to conduct soil tests before any planting activity and make recommendation for fertilizer use. Following this sustainable approach, farmers can optimize their fertilizer use, prevent soil degradation and obtain a higher return from the farm.

The emission of CH_4 from submerged paddy fields was also a major contributor to the GHG in the PP and PW systems (12.2 and 18.6%, respectively). In certain parts of the country (especially southern India), the System of Rice Intensification (SRI) method is followed for paddy-rice cultivation. This system uses lesser amount of water compared to the conventional paddy-rice production in which the field is continuously flooded. If the farmers adopt such system, they would lower the GHG emission pertinent to anaerobic decomposition of organic material in submerged paddy-rice field, and on-farm fossil fuel and machinery use. In addition, they would also lower the input cost by reducing the use of fuel in irrigation systems as well as resources for managing weeds in their respective crop production systems.

4. Conclusions

Paddy rice–based cropping systems are most followed cropping systems in the Middle Indo–Gangetic plains of Indian subcontinent. Due to flooded paddy rice fields and extensive use of fertilizers, agrochemicals and mechanized means of farming, the paddy rice–based systems seem to be a major source of GHG emissions in the form of CO_2, CH_4 and N_2O. Following conclusions can be drawn based on the current study for the assessment of GHG emission from the two cropping systems (PP and PW) in the study area.

- The two systems are different in terms of total GHG emission, with the PP system having a significantly higher GHG emission than the PW system, owing to a higher use of agricultural inputs in the production processes.
- GHG emission due to fertilizer use (during production and on–farm emissions) was the major contributor to the GHG pool in both the systems, followed by direct N_2O emissions and on farm emissions due to the use of fuel and electricity.
- While comparatively larger farms accounted for higher GHG emission, the net return was also higher for such farms which conforms the economy of scale.
- There existed a trade-off between GHG emission and net return for different farm sizes in PW system, however, comparatively larger farms were found more eco–efficient compared to the smaller farms. In PP system, different farm sizes had numerically similar EE values.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgement

This activity was a part of Masters study of the first author, who received scholarship from the Asian Institute of Technology, Thailand. Authors would like to express their sincere gratitude to Mr. Shankar Kumar for his effort in coordinating with the farmers and scheduling the interviews. Authors would also like to acknowledge the farmers of the study area for their valuable time for answering the questionnaire patiently and accurately.

References

Ahmad, S., Li, C., Dai, G., Zhang, M., Wang, J., Pan, S., Cao, C., 2009. Greenhouse gas emission from direct seeding paddy field under different rice tillage systems in central China. Soil Tillage Res. 106, 54–61.

Astier, M., Merlín-Uribe, Y., Villamil-Echeverri, L., Garciarreal, A., Gavito, M.E., Masera, O.R., 2014. Energy balance and greenhouse gas emissions in organic and conventional avocado orchards in Mexico. Ecol. Indicat. 43, 281–287.

Bhatia, A., Pathak, H., Jain, N., Singh, P.K., Singh, A.K., 2005. Global warming potential of manure amended soils under rice–wheat system in the Indo–Gangetic plains. Atmos. Environ. 39 (37), 6976–6984.

Bhatia, A., Aggarwal, P.K., Jain, N., Pathak, H., 2012. Greenhouse gas emission from rice– and wheat–growing areas in India: spatial analysis and upscaling. Greenh. Gases. 2, 115–125.

Carbon Footprint, 2019. Country specific electricity grid greenhouse gas emission factors. Available at: https://www.carbonfootprint.com/docs/2019_06_emissions_factors_sources_for_2019_electricity.ppd. Accessed 30 May 2020.

Dalal, R.C., Wang, W., Robertson, P., Parkg, W.J., 2003. Nitrous oxide emission from Australian agriculture lands and mitigation options: a review. Aust. J. Soil Res. 41, 165–195.

FAOSTAT, 2019. Available at: http://www.fao.org/faostat/en/#data/GT. Accessed 14 December 2019.

Gomez–Limo, J.A., Picazo–Tadeo, A.J., Reig, M., 2012. Eco–efficiency assessment of olive farms in Andalusia. Land Use Pol. 29 (2), 395–406.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories for Agriculture, Forestry and Other Land Use. The Institute for Global Environmental Strategies (IGES), Japan.

IPCC, 2019. 2019 refinement to the 2006 IPCC guidelines for national greenhouse gas inventories. Available at: https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/. . Accessed 06 June 2020.

Kggwane, M., Mdlambuzi, T., Moeletsi, M., Tsubo, M., Mliswa, V., Grootboom, L., 2016. Greenhouse gas emissions from different crop production and management practices in South Africa. Environ. Dev. 19, 23–35.

Khoshnevisan, B., Rafiee, S., Omi, M., Yousefi, M., 2013a. Modeling of energy consumption and GHG (greenhouse gas) emissions in wheat production in Esfahan province of Iran using artificial neural network. Energy 52, 333–338.

Khoshnevisan, B., Rafiee, S., Omid, M., Mousazadeh, H., 2013b. Comparison of GHG emissions of efficient and inefficient potato producers based on data envelopment analysis. J. Agri. Engg. Biotech. 1, 81–88.

Khoshnevisan, B., Rafiee, S., Omid, M., Mousazadeh, H., 2013c. Applying data envelopment analysis approach to improve energy efficiency and reduce GHG (greenhouse gas) emission of wheat production. Energy 58, 588–593.

Koga, N., Tajima, R., 2011. Assessing energy efficiencies and greenhouse gas emissions under bioethanol–oriented paddy rice production in northern Japan. J. Environ. Manag. 92, 967–973.

Kourous, G., 2014. Agriculture's greenhouse gas emissions on the rise. Available at: http://www.fao.org/news/story/en/item/216137/icode/. . Accessed 28 January 2020.

Kushwah, V.S., Singh, S.P., 2011. Relative performance of low input and high input technology for potato production in India. Potato J. 38 (1), 56–60.

Lal, R., 2004. Carbon emission from farm operation. Environ. Int. 30 (7), 981–990.

Linquist, B., van Groenigen, K.J., Adviento–Borbe, M.A., Pittelkow, C., van Kessel, C., 2011. An agronomic assessment of greenhouse gas emissions from major cereal crops. Global Change Biol. 18 (1), 194–209.

Ma, Y.C., Kong, X.W., Yang, B., Zhang, X.L., Yan, X.Y., Xiong, Z.Q., 2013. Net global warming potential and greenhouse gas intensity of annual rice–wheat rotations with integrated soil–crop system management. Agric. Ecosyst. Environ. 164, 209–219.

Maraseni, T.N., Cockfield, G., 2011. Does the adoption of zero tillage reduce greenhouse gas emissions? An assessment for the grains industry in Australia. Agric. Syst. 104, 451–458.

Maraseni, T.N., Cockfield, G., Apan, A., 2007. A comparison of greenhouse gas emissions from inputs into farm enterprises in Southeast Queensland, Australia. J. Environ. Sci. Health. Part A. 42, 11–19.

Muller, K., Sturm, A., 2001. Standardized Eco–Efficiency Indicators. A concept paper 2001. Switzerland.

Pandey, D., Agrawal, M., Bohra, J.S., 2013. Impact of four tillage permutations in rice–wheat system on GHG performance of wheat cultivation through carbon foot printing. Ecol. Eng. 60, 261–270.

Pathak, H., Bhatia, A., Prasad, S., Singh, S., Kumar, S., Jain, M.C., Kumar, U., 2002. Emission of nitrous oxide from soil in rice–wheat systems of Indo–Gangetic plains of India. J. Environ. Monitor Assess. 77 (2), 163–178.

Pishgar–Komleh, S.H., Ghahderijani, M., Sefeedpari, P., 2012. Energy consumption and CO_2 emission analysis of potato production based on different farm size levels in Iran. J. Clean. Prod. 33, 183–191.

Pratibha, G., Srinivas, I., Rao, K.V., Raju, B.M.K., Thyagraj, C.R., Korwar, G.R., Venkateswarlu, B., Shanker, A.K., Choudhary, D.K., Rao, K.S., Srinivasrao, C., 2015. Impact of conversion agriculture practices on energy use efficiency and global warming potential in rainfed peigonpea–castor systems. Eur. J. Agron. 66, 30–40.

Rab, M.A., Fisher, P.D., O'Halloran, N.J., 2008. Preliminary estimation of the carbon footprint of the Australian vegetable industry. Discuss. Pap. 4, 37p.

Saling, P., Kicherer, A., Kramer, B., Wittlinger, R., Zombik, W., Schmidt, I., Schrot, S., Schmidt, W., 2002. Eco–efficiency analysis by BASF: the method. Int. J. Life Cycle Assess. 7, 203–218.

Šarauskis, E., Masilionytė, L., Juknevičius, D., Buragienė, S., Kriaučiūnienė, Z., 2019. Energy use efficiency, GHG emissions, and cost-effectiveness of organic and sustainable fertilization. Energy 172, 1151–1160.

Seguin, B., Arrouays, D., Balesdent, J., Soussana, J., Bondeau, A., Smith, P., Zaehle, S., de Noblet, N., Viovy, N., 2007. Moderating the impact of agriculture on climate. Agric. For. Meteorol. 142, 278–287.

Sinha, R., Ranjan, R., Khot, L.R., Hoheisel, G.-A., Grieshop, M.J., 2019. Drift potential from a solid set canopy delivery system and an axial–fan air–assisted sprayer during applications in grapevines. Biosyst. Eng. 188, 207–216.

Soltani, A., Rajabi, M.H., Zeinali, E., Soltani, E., 2013. Energy inputs and greenhouse gases emissions in wheat production in Gorgan, Iran. Energy 50, 54–61.

Soni, P., Taewichit, C., Salokhe, V.M., 2013. Energy consumption and CO_2 emissions in rainfed agricultural production systems of Northeast Thailand. Agric. Syst. 116, 25–36.

Soni, P., Sinha, R., Perret, S.R., 2018. Energy use and efficiency in selected rice–based cropping systems of the Middle–Indo Gangetic Plains in India. Energy Rep. 4, 554–564.

Statistical Year Book, India, 2012. Available at: http://mospi.nic.in/statistical–year–book–india/2012/. Accessed 15 October, 2016.

World Bank, 2008. Agriculture for development. Available at: https://siteresources.world.bank.org/INTWDR2008/Resources/WDR_00_book.pdf. Accessed 12 January, 2017.

Xue, J.–F., Yuan, Y.–Q., Zhang, H.–L., Ren, A.–X., Lin, W., Sun, M., Gao, Z.–Q., Sun, D.–S., 2018. Carbon footprint of dryland winter wheat under film mulching during summer–fallow season and sowing method on the Loess Plateau. Ecol. Indicat. 95 (1), 12–20.

Yao, Z., Zheng, X., Xie, B., Liu, C., Mei, B., Dong, H., Butterbach–Bahl, K., Zhu, J., 2009. Comparison of manual and automated chambers for field measurements. Atmos. Environ. 43, 1888–1896.

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

Zhang, X.–Q., Pu, C., Zhao, X., Xue, J.–F., Zhang, R., Nie, Z.–J., Chen, F., Lal, R., Zhang, H.–L., 2016. Tillage effects on carbon footprint and ecosystem services of climate regulation in a winter wheat–summer maize cropping system of the North China Plain. Ecol. Indicat. 67, 821–829.