A Life Cycle Assessment (LCA) of Greenhouse Gas Emissions from SRI and Flooded Rice Production in SE India

東南印度水稻強化栽培系統與傳統水耕稻作溫室氣體排放量之生命週期評估

Alfred Gathorne-Hardy[1*], with D. Narasimha Reddy[2],
Motkuri Venkatanarayana[2] and Barbara Harriss-White[1]

[1]Oxford University, Oxford, UK; [2]National Institute for Rural Development, Hyderabad, India

ABSTRACT

Rice feeds more people than any other crop, but each kilogram of rice is responsible for substantially more greenhouse gas (GHG) emissions than other key staple foods. The System of Rice Intensification (SRI) has recently received considerable attention for its ability to increase yields while using less water. Yet so far there has been little research into the GHG emissions associated with SRI production systems, and how they compare to those from conventional flooded-rice production techniques.

A streamlined Life Cycle Assessment (LCA) methodology was used to compare the GHG emissions and groundwater use from SRI and from conventional rice production. Input data were derived from farmer questionnaires in SE India and appropriate secondary data sources.

The results showed that SRI methods substantially raised farmers' yields, from 4.8 tons to 7.6 ton per hectare, a 58% increase, while reducing water applications. At the same time it was seen that SRI management offered opportunities for significant GHG reductions, both per hectare and per kilogram of rice produced. These savings principally arise from reduced methane emissions and reduced embodied emissions in the electricity used to pump water for irrigation. SRI nitrous oxide emissions were somewhat higher than on control farms, but the difference was significant only per hectare, not per kg of rice. The net effects of SRI practice on reducing global warming potential were positive in that the small increases in N_2O did not offset the larger diminishment of CH_4.

Keywords: flooded rice production, greenhouse gas emissions, Life Cycle Assessment, System of Rice Intensification.

摘　　　要

稻米為多數人的主食，但一公斤稻米產量所排放之溫室氣體卻遠多於其他糧食作物。水稻強化栽培系統使用較少之用水量增加作物產量，已引起廣泛注意。然而，目前對於溫室氣體排放量與水稻強化栽培系統之關聯性，及此系統與傳統水耕稻作技術之比較等研究仍相當缺乏。

本研究採用簡約式生命周期評估方法，來比較水稻強化栽培系統與傳統水耕稻作間溫

[*] Corresponding author : alfred.gathorne-hardy@area.ox.ac.uk

室氣體排放量與地下水使用量之差異性。分析上所需的樣本資料為東南印度的農民問卷及次級資料。

分析結果顯示，水稻強化栽培系統大幅提升作物產量而且降低用水量，作物產量由每公頃4.8噸增加至每公頃7.6噸，增加幅度為58%。同時，水稻強化栽培系統對於每公頃每公斤稻米產量所產生的溫室氣體排放量有顯著改善，主要因素為甲烷排放量與抽取灌溉用水所需電力減少。此外，每公頃(非每公斤)的氮氧化物排放量在水稻強化栽培系統中略高於傳統水耕稻作。整體而言，甲烷(CH_4)的降幅程度遠大於一氧化二(N_2O)的小幅增加量，水稻強化栽培系統對於減緩全球暖化效應是有助益的。

關鍵詞：水耕稻作，溫室氣體排放量，生命週期評估，水稻強化栽培系統。

1.0 INTRODUCTION

Rice is the most important staple crop worldwide, directly feeding more than half the world's population. However, rice is also the most greenhouse gas-intensive staple crop, producing about four times more GHG emissions per ton than wheat or maize (Linquist et al. 2012). The System of Rice Intensification (SRI) is novel set of rice production techniques that has recently received considerable attention for their ability to increase yields. This paper uses a streamlined Life Cycle Assessment (LCA) methodology (White 2010) to compare the GHG emissions and groundwater use resulting from conventional and SRI rice production systems.

Three GHGs are important from rice production: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). GHG emissions from conventional rice production are largely associated with its water use via two mechanisms; first, through the embodied emissions associated with irrigation provision, and second, from flooded soils which create the anaerobic soil environments that drive microbial methane production. While rice does not need to grow in flooded conditions, most rice does come from ponded systems (IRRI 2013). This production system uses an estimated 34-43% of global irrigated water, or 24-30% of the total freshwater withdrawals (Bouman et al. 2007). The pattern of GHG emissions from non-flooded rice can be very different, with nitrous oxide playing an important role.

Embodied irrigation emissions derive from the energy used for pumping water. In gravity-fed irrigation systems, this energy requirement can be minimal; but it can be high with diesel-based groundwater extraction systems, or when using electricity not generated by hydropower (Nelson et al. 2009).

The net methane emission from any specific paddy field is determined by a wide range of factors (Adhya et al. 2000; Gathorne-Hardy 2013a; Yan et al. 2005), but two main drivers dominate methane production: irrigation patterns, and the quantity of organic material in the soil. The rate of oxygen diffusion through water is 10,000 times slower than through air, so irrigation with flooding/ponding produces anaerobic conditions within hours of flooding (Adhya et al. 2000; Bodelier 2003; Chanton et al. 1997).

Under anaerobic soil conditions, methanogens (methane-producing bacteria) produce methane as a by-product of their respiration of organic matter. Under such conditions, the supply of substrates for methanogens' subsistence is the commonest limiting factor (Wang et al. 2000; Yao et al. 1999). Organic matter in paddy fields originates from both direct by-products of rice production (such as sloughed-off root cells and root exudates) and from added materials (manures and crop residues). Control of methanogens' substrates is only possible from modifying the latter.

In most arable agriculture, N_2O is the dominant GHG, for example, responsible for 80% of GHG emissions from wheat production (Woods et al.

2008). Nitrous oxide is naturally produced from soils during nitrification and de-nitrification by bacteria. In India, N_2O is estimated to be responsible for 13% of overall agricultural GHG emissions (MEF 2010).

However, direct N_2O emissions from flooded paddy fields are minimal, as nitrification does not occur under anaerobic conditions. Consequently neither can de-nitrification occur, due to lack of NO_3^- in the soil (Qin et al. 2010). Even though minimal N_2O emissions are likely from the flooded soils, still some off-site (indirect) N_2O emissions are likely from irrigated rice production due to the addition of nitrogen fertilizer to fields because nitrogen compounds can leach through the soil or are directly drained from the site.

The frequent draining of SRI-managed paddies creates intermittently aerobic soil conditions that will allow bacterial nitrogen cycling in the soil, which can potentially increase the N_2O emissions from SRI fields compared to flooded fields (Hansen et al. 1993; Yan et al. 2000; Zheng et al. 2000). A trade-off is therefore expected between methane and nitrous oxide as fully anaerobic soils inhibit the generation of N_2O while facilitating methane emissions. A more complicated relationship between CH_4 and N_2O emissions with regard to flooded paddy fields is reported in this issue, however, when soil temperature, pH and other factors are taken into account (Setiawan et al.). Also, several unpublished studies of this relationship have not found N_2O increases offsetting NH_4 reductions under SRI management. So this is a subject warranting further detailed investigation.

1.1 The System of Rice Intensification

SRI methodology, developed in Madagascar in the 1980s, involves certain basic practices that differ from age-old production techniques: (a) earlier transplanting, at the 2-3 leaf stage, typically less than 15 days old, compared to up to 30-45 days old for conventional rice transplanting; (b) only one

plant per hill, in contrast to 3-5 or more plants; (c) hills laid out in a grid pattern with wide spacing; (d) increased use of manure and other organic fertilizers; and (e) intermittent flooding and draining of the rice paddies rather than continuously flooded soils (Reddy and Mortkori 2013; Stoop et al. 2002; Uphoff 2008). Controlling weeds by the use of a mechanical implement which disturbs the soil surface is highly recommended to achieve the active soil aeration that better supports the growth of roots and the aerobic soil biota.

While there has been controversy in the literature about the range of benefits associated with SRI (Sumberg et al. 2013), a wide range of scientific journal articles, NGO literature, and anecdotal reports suggest that grain yield gains are usually associated with it, as well as reduced water use (Kassam et al. 2011; Sinha and Talati 2007; Uphoff 2008; WWF 2007). However, there has been very little measurement of GHG emissions from SRI fields compared to conventional paddy production. The literature offers little evidence that can used to assess reliably the relative GHG impact of SRI crop and water management compared to conventional production techniques.

Without systematic and quantified evidence it is hard to predict what the GHG impact is likely to be since SRI management has direct impacts on many of the key paddy GHG production processes. Methane emissions may be increased due to increased organic soil amendments, or they could be decreased due to increased soil aeration. Nitrous oxide (N_2O) emissions, usually minimal from anaerobic (flooded) paddy fields, may increase when soils are made more aerobic under SRI practice; or they could be little changed if the N supplied to fields is in organic rather than inorganic forms, so there is little excess reactive N to be converted by microbial activity. Repeated flooding and drainage should result in less water use; but whether pumping is actually reduced will depend upon the actual amount of flooding and drainage compared to

traditional water use, as well as on the kind and amount of energy used for irrigation.

1.2 Life Cycle Assessment

The research reported here modelled and evaluated GHG emissions (CO_2, CH_4 and N_2O) from SRI compared to conventional flooded-irrigation paddy production techniques up to the farm gate. Data were collected from Andhra Pradesh, India. It is part of a larger project that generates and utilizes primary data to understand how a range of environmental effects (GHG emissions, fossil and non-fossil energy, groundwater), social considerations (e.g., the qualities, quantities and gender of jobs), and economic factors (costs, returns, value-added) interact along the entire rice production-distribution supply chain, from field to shop, for four different rice production techniques (SRI, organic, rainfed, and current conventional management) and four different retail outlets. More information can be found at http://www. southasia.ox.ac.uk/research-resources-potential-new-applications-resources-greenhouse-gases-technology-and-jobs-indias.

GHG emissions from rice production were quantified using an analytical tool called Life Cycle Assessment (LCA). LCA is defined as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006). This includes all stages required for the creation of the material of interest through to its disposal or recycling, and it includes a variety of criteria that range from energy use to eco-toxicity. The research detailed in this paper is limited to assessing GHG emissions and groundwater use, not the wider range of environmental impacts included within a full LCA, so accordingly we refer to the analysis as "streamlined" LCA (White 2010).

1.2.1 Issues of allocation

When more than one output is produced from a process, the associated emissions must be allocated between or among them (European Commission 2010; ISO2006). Consider, for example, the role of livestock in paddy production. In addition to providing manure for maintaining soil fertility, and in some cases providing traction (ploughing), cows produce calves and milk. All of these outputs may have economic value. To assign all the emissions from cows to just one of these outputs would give one output an inflated debit and would give the other outputs unwarranted credit in terms of their respective contributions to global warming potential (GWP).

Accordingly, the overall emissions from a production process must be allocated appropriately between or among the outputs. Allocation can occur through different measures, including by mass, by energy content, or by value (Anon. 2009). Economically-based allocation of emission 'debits' best reflects the drivers of production (Williams et al. 2006). In economic allocation, the proportional allocation of emissions to each co-product is determined by the respective economic value of each co-product, on the assumption that the co-products drive the total production in proportion to their economic value.

The paddy field itself is a multi-functional resource, producing straw as well as paddy. Yet in rice LCAs, rice straw is commonly not allocated to the emissions attributed to rice (Blengini and Busto 2009), or even more commonly it is not even mentioned (Blengini and Busto 2009; Hokazono and Hayashi 2012; Wang et al. 2010). As a useful by-product (straw can be used for animal bedding, as soil mulch cover, for fodder or cooking, and in some parts of India for paper making), straw should be allocated a share of the environmental burden associated with rice production.

Since approximately equal weight and energy are involved in the production of straw and paddy grain, the environmental costs/benefits of straw and paddy should be allocated equally if mass or

energy is used as the criterion of allocation. But this would not reflect the motivating factors for paddy farming, which is driven mostly by paddy production expectations, with straw considered by producers as a potentially useful by-product. This makes economically-based allocation more realistic. Economic methods to allocate emissions to straw are discussed under Methods, below.

2.0 METHODS

This research modelled GHG emissions from SRI and conventional/intensive rice production techniques (the latter hereafter referred to as 'control') from a case study site in Andhra Pradesh, India. A methodology known as Life Cycle Assessment (LCA) was used to determine the GHG emissions based on the standards and criteria of ISO 14040, PAS 2050, and the ILCD handbook (European Commission 2010; ISO 2006; BSI 2011). A mixture of primary and secondary data was involved to construct the quantified analysis.

Primary data on farmers' practises and results were collected in the second half of 2012 for the crops of the preceding agricultural year (2011-2012). Data were collected using recall methods via an extensive (31-page) survey of selected farm households. These data were collected as part of a wider project, looking at how social, economic and environmental criteria all interact within the rice production and supply chain. Secondary data included inventory data from a range of sources, as detailed in the Annex.

Data collection took place in the Janagaon region of Warangal District, Andhra Pradesh, SE India. This is a semi-arid area with average annual rainfall of 865 mm, concentrated in the SW monsoon season, with 624 mm falling from June to September (Anon. 2013). The altitude is approximately 380 m above sea level.

Twenty 'SRI' farms were sampled along with 10 control farms in the area. While this sample size is not large due to the logistical limitations associated with the extensive data collection required for the wider project as discussed in section 1.2 above, the results reviewed in section 3.0 passed a number of statistical tests and thus are unlikely to be due just to chance. The control farms were ones using standard, intensive, flooded paddy production and 'Green Revolution' production methods. The key agronomic practices used in the two sets of farmers are reviewed in sections 3.1 and 3.2 below.

The farms using SRI methods in our sample were not necessarily using the full set of SRI practices or using them all as recommended, but there were demonstrable and significant differences between the sets of practices being evaluated. Possibly the differences in results reported here would have been greater or different if the evaluation of SRI methods had been done more strictly, under strictly controlled conditions. But for this study, we wanted to consider the results of actual practice, which is invariably imperfect. In philosophical terms, what was evaluated was the 'existence' of SRI rather than its 'essence.'

Individual farms for both control and SRI surveys were chosen using a semi-random snowballing technique, the semi-random aspect including selection of farms aimed to ensure that the distribution of farms sizes was representative of that in the area as reported in the Agricultural Census. There proved to be no statistical difference between control and SRI farms in key socio-economic and environmental attributes (farm size and water table), as seen in section 3.2. Climate variables were not considered due to the close and overlapping locations of the control and SRI farms.

2.1 Functional units

This paper uses two approaches to understand the GHG impacts of SRI production techniques – a bottom-up, land-based approach, and a top-down, product-based approach. Bottom-up approaches are useful for measuring those environmental impacts

that are most relevant (directly or indirectly) on an area basis – for example, groundwater use, electricity use, as well as impacts not measured in this study (water pollution, local air pollution, etc.). Bottom-up, area-based measures are also the metrics most commonly used by farmers. However, consumers do not buy areas of land; they buy the produce from the land. From their perspective, environmental impact per unit of product (paddy rice) is a relevant metric.

Two functional units for analysis are thus used to reflect these bottom-up and top-down approaches to evaluating rice production, respectively:

a. 1 hectare for one season of paddy production (area-based)

b. 1 kg of paddy at the farm gate (product-based).

2.2 System baselines and boundaries

Establishing an appropriate baseline (counter-factual) and boundaries is critical for accurate, meaningful and representative results (Gathorne-Hardy 2013b). In this study, the baseline to which SRI practices are compared is the Green Revolution paddy-farming system by which most rice is produced in India today.

The boundaries included all inputs and processes necessary to grow paddy that make a material difference to the final figures. We used 1% of the total as our criterion for a material difference and did not consider any effect that was less than this. In this paper, inputs to post-harvest processes that convert paddy into processed, polished rice were not included. These are covered elsewhere (Gathorne-Hardy and Harriss-White 2013).

The boundaries to this study are shown in Figure 1. The central black box shows the processes for which we collected data. Everything within the inside box is included in the study. Outside this box are four key elements that have not been included in the research reported here.

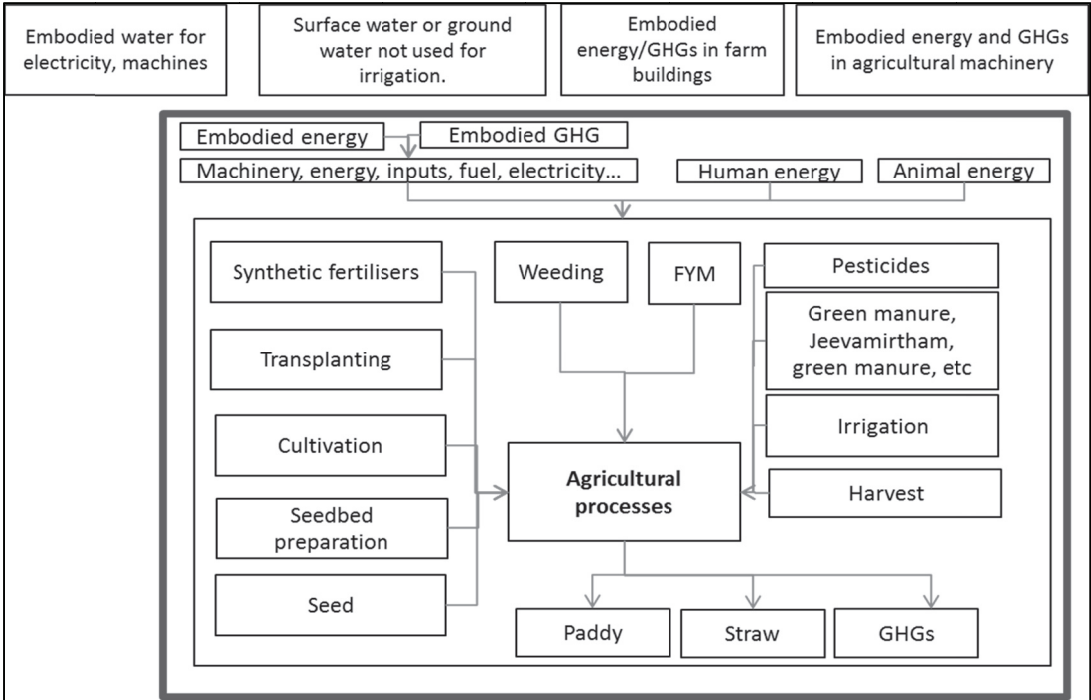Embodied water (the water used off-site for the production of goods/energy, for example, tractors

Figure 1. System boundaries for determining the environmental burden of rice production. Categories outside the heavy line are not included within the boundary of this LCA.

or electricity) was not included in the analysis due to lack of available data. Neither was surface water included due to lack of ability to capture such data accurately using recall surveys. However, this source of water contributes practically nothing to GWP, and no farms in the sample irrigated with any other sources of water except groundwater and precipitation.

Embodied emissions associated with buildings were also not included because paddy is normally stored – if stored at all – in the producer's dwelling, and no other buildings were deemed relevant to the production system for the field conditions observed. The main items of machinery such as tractors and power tillers are included. Early analysis showed that the combined embodied GHG emissions for other implements (weeders, trailers, levelling plates, and ploughs) fell well below the 1% inclusion criterion for material difference; so these were not included in the analysis.

Allocation between co-products has been based on economic methods in this study. Allocation to straw is made more difficult within this study for three reasons: (i) there is rarely a market for the straw, (ii) few farmers know how much straw they harvest, and (iii) straw that leaves the field sometimes, although not always, is returned to the field as manure. The methods and measures that we have used to allocate GHG emissions to straw are detailed in the next section.

2.3 Inventory analysis and data sources

This section explains what information was collected, and what conversion factors were used to generate the LCA model. This model aims to account for the amounts and sources of GHG emissions that drive up global warming potentials, discussed below in some detail.

2.3.1 Assessment of global warming potentials

To calculate GHG equivalents, we used 100-year global warming potentials (GWP100) as specified by IPCC 2007 (Forster et al. 2007). GWP is a measure of how much heat is retained in the atmosphere for each gas, and the 100 indicates that the figures have been averaged to assess the heat retained over a period of 100 years. GWPs are set up in reference to carbon dioxide (CO_2), which is given a value of 1.

The three main agricultural GHGs of concern are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), which have GWPs, respectively, of 1, 25 and 298, meaning that over 100 years, 1 kg of N_2O released to the atmosphere will retain 298 times the amount of heat retained by 1 kg of CO_2 in the atmosphere. Using this metric, 1 kg of CO_2 is 25 times less carbon-polluting than releasing 1 kg of CH_4. Levels of CH_4 and N_2O emissions for flooded control irrigation paddy farm systems were taken directly from IPCC (2006). Emissions for SRI rice production systems are not included in the IPCC report, so measured emissions from controlled irrigation trials were used as a proxy, as described in the Annex. The data sources and assumptions or coefficients used in calculation are given in the Annex.

2.3.2 Inputs not covered in the analysis

Jeevamirtham, a solution of water, manure, cattle urine, jaggery and various other products, is used by organic and non-organic farmers as a fertilizer and pesticide. Typically it is mixed and retained in a barrel for a few days before it is applied to the crop, often with the incoming water. The nutrient content of its ingredients is not substantial, but this mixture has been reported to provide additional N for the soil and crop via N-fixing bacteria. This input has not been included, but is not large.

Azospirillum, phosphobacteria, Pseudomonas, Azolla, and other microbial/biological products used with SRI production have not been assigned embodied emissions in this analysis. There are no data on which to base any assumptions about their

GHG emissions, and when the authors inspected the facilities where they are produced, there was very little capital infrastructure involved. Any embodied emissions in these products are likely to be minimal, well below the 1% criterion.

2.3.3 Analysis of data

Analysis was carried out using a LCA model built in Excel, and statistical findings were tested with SPSS software.

3.0 RESULTS AND DISCUSSION

3.1 SRI practices in the study site

The SRI principles as discussed in the introduction were not necessarily always precisely followed on the ground. For example, the average age at transplanting was somewhat greater than 15 days, although about half that of conventional practice; and typically more than 1 plant was transplanted per hill (Table 1). Yet with the exception of farm yard manure applications, measures on the SRI farms studied were significantly different, and in the expected direction, compared to those on control farms. The results reported are thus not for the full use of recommended SRI practices. Such a pattern of partial and adaptive adoption of new technology and practices is a widely observed phenomenon, characteristic of the early stages of waves of most agricultural innovation (Richards 1985).

3.2 SRI yield and GHG emissions compared to control farms

SRI management was found to increase grain yield by approximately 60% compared to the control farms, 7.6 compared to 4.8 t ha^{-1}, respectively, $p < 0.001$) (Table 2a). This is a remarkable gain in yield when we consider that ambitious breeding programmes typically provide yield gains of less than 1% yr^{-1} (Fischer and Edmeades 2010). With increasing resource scarcity, including that of land, increasing yields from the same level of inputs is likely to result in reduced adverse environmental impacts.

Yet yield gains alone are not enough to guarantee environmental improvements; yield gains from the increased use of certain inputs can be associated with local environmental losses. However, in the case of SRI, the higher yield compared to the control farms was associated with lower levels of inputs – significantly lower with respect to synthetic N, P and K, and water (Table 2b). The exception to this was a slightly higher application rate of FYM on the SRI farms (18.7 compared to 17.0, $p > 0.1$). This, however, resulted in no significant difference between the total applications of N per hectare between the two production systems.

This strongly suggests that there is an inherent yield advantage in the SRI farming system compared to the traditional wetland paddy system. To which aspect of SRI this may be attributed is not yet clear – wider spacing, earlier transplanting, single plants per hill (all of which contribute to increased tillering), or to the change in irrigation patterns. Ascertaining this will require further factorial trials.

Table 1. Agronomic practices of SRI and control farmers

Agronomic practices	SRI	Control (conventional Green Revolution techniques)
Planting date (age of seedling, in days)	18 (1.1)	32.5 (1.7)**
Number of plants per hill	1.8 (0.2)	5 (0)***
Distance between hills (cm)	25 (0)	12 (0)***
Number of days irrigation (days/season)	71.8 (6.2)	118.8 (12.8)***
Quantity of manure (t ha-1)	18.7 (4)	17.0 (2)

No asterisk = no sig difference, *= p < 0.1, **= p < 0.05, ***p < 0.01. (S.E. in parentheses)

Table 2a. Farm details, GHG emissions, yield, and water use from SRI and control farms*

Farming system	Number of farms	Farm size (ha)	GHG emissions (CO_2-eq ha^{-1})	GHG emissions (kg CO_2-eq kg paddy^{-1})	Yield (kg ha^{-1})	Water use (t ha^{-1})	Water use (t kg paddy^{-1})
SRI	20	1.23 (0.62)	10,232*** (588)	1.1*** (0.05)	7,609*** (304)	16,049** (3,682)	2.05*** (0.47)
Control	10	1.11 (0.33)	13,981 (696)	2.8 (0.2)	4,834 (197)	24,980 (3,274)	4.90 (0.46)

*Asterisks indicate degree of significant difference between the farming systems; no asterisk = no significant difference, *= $p < 0.1$, **= $p < 0.05$, ***$p < 0.01$ (S.E. shown in parentheses).

Table 2b. Nutrient inputs and water table for SRI and control farms*

Farming system	Synthetic N (kg N ha^{-1})	Total N (kg N)	N used by crop (kg ha^{-1})	NUE (kg N utilised kg^{-1} N applied)	Total synthetic K	Total synthetic P	Water table (m)
SRI	157** (36)	436 (100)	129 (30)	34%** (8)	80* (18)	166** (38)	27.01 (7.30)
Control	226 (17)	478 (42)	82 (3)	18% (1)	111 (18)	250 (25)	27.28 (3.27)

*Asterisks indicate degree of significant difference between the farming systems; no asterisk = no significant difference, *= $p < 0.1$, **= $p < 0.05$, ***$p < 0.01$ (S.E. shown in parentheses).

Table 2a shows the reduced GHG emissions per hectare-season with SRI management. The emission savings are over 25%: 13,981 to 10,232 kg CO_2-eq ha^{-1} from the control and SRI fields, respectively ($p < 0.01$). The lower emissions from SRI are magnified when evaluated in terms of GHG per kg of paddy, due to the higher yield from SRI. Thus, SRI is responsible for less than half as much GHG emission per kilogram ($p < 0.001$) as with conventional rice production.

When emissions are analysed by farm operations, the pattern of emissions differs between the two systems. SRI emissions per hectare are higher for both N_2O ($p < 0.01$) and transport of manure ($p > 0.1$), but they are lower for all other categories, including significantly lower CH_4 ($p < 0.01$) and irrigation-based ($p < 0.01$) emissions (Figure 2).

Overall GHG emissions per kg of paddy show a similar pattern as above, but due to the higher yields with SRI management, N_2O emissions per kg of SRI paddy rice were not significantly different from the control N_2O emissions per kilogram, while the differences between SRI and control in terms of their CH_4 and irrigation-based emissions per kg of paddy increase (Figure 3).

While significantly less methane is emitted per hectare with SRI compared to the control farming systems, methane is still a major source of GHG from SRI production. It is responsible for almost the same proportion of total GHG emissions in comparison to the controls (47%), even though with SRI the soil is not consistently flooded. One reason for the SRI methane emissions is the higher application rates of organic amendments which serve as substrate for methanogens. This does not imply that organic soil amendments should be reduced in SRI farms; they could be an important factor in increasing SRI yields. Research into optimum organic application rates should be undertaken so that excessive rates are not used. This is likely to become increasingly important with reduced livestock stocking densities in rural areas reducing the availability of manure.
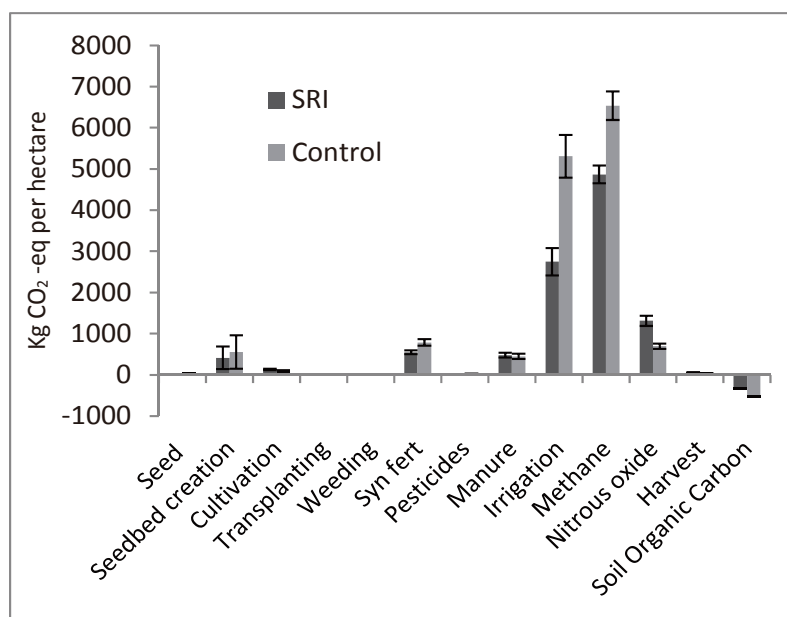
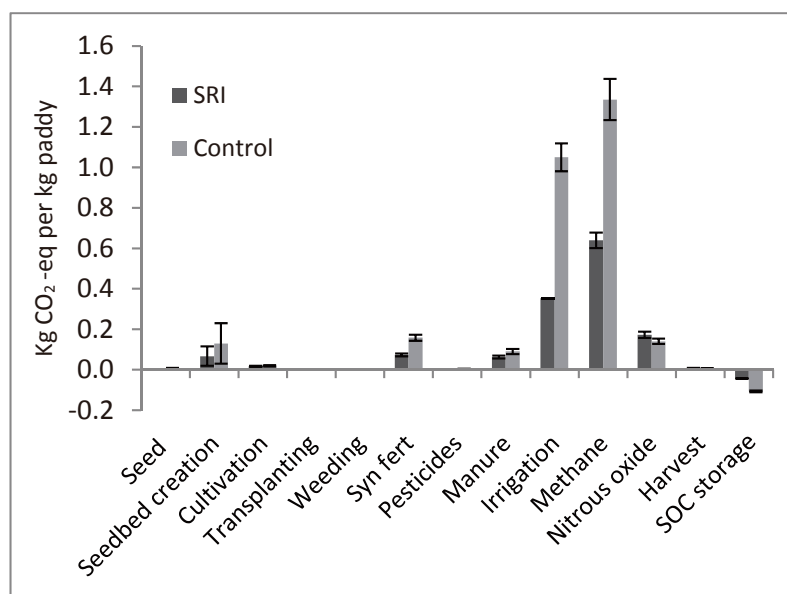Figure 2. GHG emissions per hectare for SRI and control paddy production techniques (error bars = 1 S.E.)

Figure 3. GHG emissions, per kg of paddy at the farm gate for SRI and control (error bars = 1 S.E.)

The modelled GHG emissions from SRI are highly sensitive to the assumed rate of methane and nitrous oxide emissions compared to the default emission factors posted by IPCC 2006. At present, the emission factors are based on three pieces of research. As more research is carried out specifically into the GHG emissions from SRI compared to control production systems, the accuracy of comparative LCA models will increase.

The magnitude of the different results modelled

above suggests that SRI will consistently show GHG savings per kilogram of rice produced compared to conventional flooded production systems. While an increase in N_2O emissions could mitigate CH_4 emission savings on an area basis, we saw increased yield and reduced irrigation demand with SRI management consistently reducing its GHG burden per kilogram of production.

Irrigation-based emissions are a substantial portion of the emissions from both production systems, respectively, 26% and 37% of total emissions per kg of paddy for the SRI and the control farms. We should note also that it is possible to irrigate farms with effectively no irrigation-based emissions, for example, with gravity-fed systems, or with clean power systems (local solar photovoltaic sources, or local wind-derived electricity). If we exclude irrigation-based emissions, SRI still has significantly lower GHG emissions, measured both per hectare and per kg of paddy produced ($p < 0.05$).

The higher yields associated with SRI allow greater productivity from available resources. Thus while there is no significant difference in the total N applied between to the two farming systems, the SRI system allowed rice to make better use of the available N, and to significantly increase ($p < 0.05$) the estimated nitrogen use efficiency (Table 2b).

4.0 CONCLUSIONS

SRI methods significantly increased paddy yield from 4.8 to 7.6 tons per hectare, with generally reduced or constant inputs. The only input increased with SRI compared to the control farms was farm yard manure, and this difference was not statistically significant. Achieving more rice production with reductions in greenhouse gas emissions makes the yield improvement all the more noteworthy.

SRI management offers significant GHG savings both per hectare and per kilogram of paddy produced. These savings principally arise from reduced methane emissions and reduced embodied emissions in the electricity used for pumping irrigation water. While SRI nitrous oxide emissions were somewhat higher than on control farms, the difference was only significant on a per hectare basis, and it did not offset the gains of lower GWP from the substantially-reduced emissions of methane.

This paper has not explored various other aspects of GHG emissions, for example, the role of indirect land use changes where yield gains can offer substantial GHG savings resulting from changes in land use elsewhere. Nor does the analysis here cover the wider social, economic and environmental indicators associated with SRI that are essential to fully understand the diffusion potential and impact of SRI. These other and broader issues are explored in more detail in papers listed at http://www.southasia.ox.ac.uk/resources-greenhouse-gases-technology-and-jobs-indias-informal-economy-case-rice

Annex: Input values, calculations, and data sources used in the LCA model

Operation	Reference unit	Figure	Data source
Seed	kg CO_2-eq ha^{-1}	See data source	The mean GHG for that production technique, +5% to account for handling and losses
Power tiller diesel fuel use	liters hr^{-1}	1.5	Farmer survey
GHG intensity diesel fuel	kg CO_2-eq l^{-1}	3.0168	Renewable energy directive (Anon. 2009)

Power tiller life expectancy	yrs	20	Farmer survey, probably an underestimate.
Power tiller weight	kg	515	Greavescotton (2013)
Tractor diesel fuel hr^{-1}	liters hr^{-1}	4	Farmer survey
Tractor weight	kg	1952.5	John Deere (2012); Mahindra (2012)
Embodied GHG of steel	kg CO_2-eq kg steel^{-1}	2.7	CSE (2012)
Bullocks	kg CO_2 -eq hr^{-1} for a pair of bullocks	4.47	Running and embodied emissions were calculated together for bullocks' use. It was assumed that all livestock-based cultivation was done by bullocks. Annual India-specific methane emissions were taken from Singhal et al. (2005) for indigenous bulls. Bullocks were assumed to work for 13 years (from 5 years to 18 years (from farmer survey). Total calf emissions were also taken from Singhal et al. (2005). The annual emissions plus 1/13th of calf emissions were divided by the total number of days that bullocks worked per year (as described by the owner) and 24 hours a day, then multiplied by the number of hours the animal worked day per day. Thus, if an animal worked 6 hours a day, each hour would represent 25% of the daily emissions, as the bullocks emit methane 24h day^{-1}. For rented bullocks, it was assumed that they worked 100 days a year, and 5.5 hours a day (Fuller and Aye 2012). This emission rate was reduced by 1.3% to account for the final value of the bullocks at the end of life (assumed to be Rs 5000: G. Rodrigo, 2012, pers. comm.). Assuming bullocks work 100 days/ yr, at 5.5 hours a day at Rs. 50 hr^{-1} (assumed to be rented out at Rs. 100 hr^{-1} for a pair, so Rs. 50 hr^{-1} bullocks^{-1}) for 13 years, and that a bullock is worth Rs 15,000 when purchased, then the end-of-life value is 1.3% of total value. It is further reduced by 4.1% for the CH4 allocated to manure, using economic value of NPK fertilizer in manure (1,166 Rs/ animal) compared to the potential rental value of an bullocks over a year (50 Rs. hr^{-1}, 5.5 hr day^{-1} 100 days yr^{-1})
Allocation to straw	% reduction in 'economic' yield	- 4%	Emissions were allocated to straw using an economic calculation; emissions allocated to straw were deducted from those allocated to paddy. We calculated the yield of straw by proxy, using the harvest index. Although the harvest index (the proportion of grain to total above-ground biomass) for intensive in-bred rice is approximately 50% (Islam et al. 2010; Khush 2001), this does not account for the actual availability of straw to harvest – inevitably a portion is left in the field. We have therefore assumed that the harvestable yield of straw would be 90% of total straw, so total straw yield is assumed to be 90% of paddy yield (0.5*2*90%). The value of straw was calculated using straw prices from Tamil Nadu, which averaged Rs 0.51 kg^{-1} (standard deviation 0.043) (Gathorne-Hardy and Harriss-White 2013).

Tractors, embodied emission.	kg CO_2 –eq hr^{-1}		Assumed to be 100% steel and to last for 20 years. Embodied GHG emissions for Indian steel taken from CSE (2012). Embodied emissions are calculated on an hourly basis by dividing the total embodied emissions by the fraction of hours for each job compared to total number of hours the tractor works in 20 years.
Fertilizers	kg CO_2-eq kg^{-1} of bought fertilizer, i.e., not just active ingredient	Phosphate 1.94 DAP 1.79 Urea 0.93 Complex 0.24 Potash 0.16 Neem 0.10	Urea emissions were taken from India-specific urea GHG emission factors from CSE (2009). Data for phosphate, ammonia and potash were taken from Woods and Cowie (2004). Neem cake: 0.01 kg CO_2-eq kg^{-1}. This used extraction data for jatropha from Kumar et al. (2012), multiplied by the economic values of neem oil and neem cake from Krishi Vighan Kendra (2012).
Pesticides	CO_2-eq kg active ingredient	4.921	Data taken from Elsayad et al. (2003).
Manure	kg CO_2-eq t^{-1}	0.03	Manure from cows/bullocks has associated CH_4 emissions from bovine enteric emissions. Methane emissions were allocated using economic measures. The annual emissions from a dairy cow, taken from Singhal et al. (2005), was split by the total value from milk production (taken from farmer surveys) and the total value of the NPK in the manure, using Tennakoon and Hemamala Bandara (2003). Calculated economic values and reported values for manure were similar (6 and 8%, respectively, $p > 0.05$), but economic measures were chosen due to the shortage of reported values.
Nitrogen use efficiency	%		Defined in this paper as quantity of nitrogen used by the crop compared to total nitrogen supply (Good et al. 2004). This assumes rice plants take up 17 kg N for each ton of rough rice produced through the grain, leaves and straw (quoted in Choudhury 2004).
Methane emissions	kg CO_2-eq ha^{-1}	Modelled	IPCC default figures (IPCC 2006).
Nitrous oxide emissions	kg CO_2-eq ha^{-1}	Modelled	IPCC default figures (IPCC 2006).
SRI CH_4 and N_2O emissions	kg CO_2-eq ha^{-1}	Modelled	Calculating soil-based GHG emissions from SRI rice was more complicated due to lack of appropriate IPCC default figures. In principle, the methane emissions should be low due to the repeated drainage, and nitrous oxide emissions would be high due to the partially-flooded soil conditions, but the authors could find no published evidence to confirm this. While there are some SRI GHG emission data available, none that the author could find were adequate for our use (i.e., with detailed, replicated methodologies, suitable analysis). For this reason we have relied on papers looking at controlled irrigation as an isolated factor (for methane, we used Peng et al. (2011b), Hou et al. (2012) and Suryavanshi et al. (2013) resulting in methane emissions of 57.9% of control (conventional/ intensive) production techniques; for nitrous oxide, we used Peng et al. (2011b), Hou et al. (2012) and Peng et al. (2011a), resulting in N2O emissions of 211.0% of control. This has some advantages, e.g., the data are not compounded by multiple factors which can be included in separate calculations (e.g., such as changes in levels of manure inputs), but it also has disadvantages since some specific SRI practices have no data relating to them at all (such as wider spacing between hills) and so this cannot be incorporated into the GHG calculations. This calculation is very close to the IPCC multiple-aeration figure of 52% for multiple aerations (IPCC 2006).

Electricity-based emissions from irrigation.			Calculating the amount of energy used for irrigation was difficult. Ideally we would have used meter readings, but no farmers had electricity meters. Instead we calculated the total amount of energy used through the size of the pump(s) and the number of hours they were used over the season. While pump horsepower was unambiguous, the hours of pump use were from daily estimates.
Embodied GHG emissions associated with electricity	kg CO_2 eq KWh^{-1}	1.1095	This was generated using CEA (2011) data of 0.81 at production multiplied by 27% T and E losses (Alagh 2010). Transmission losses are included, but not theft because stolen electricity presumably has some utility, so it should also take a share of overall emissions.
Harvest	GHG emissions hr^{-1}	15.084	5 liters of diesel fuel hr-1, taken from interview with combine harvester owner/operator.
Soil organic carbon	kg CO_2-eq ha^{-1}	Modelled	Taken from IPCC (IPCC 2006).

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