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Assessing the environmental impacts of irrigation and drainage systems with Life Cycle Analysis

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ASSESSING THE ENVIRONMENTAL IMPACTS OF IRRIGATION AND DRAINAGE SYSTEMS WITH LIFE CYCLE ANALYSIS

Scientific editors: SR Perret, M Van der Laan, N Hatcho

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Assessing the Environmental Impacts of Irrigation and Drainage Systems: The Potential Contribution of Life Cycle Analysis

Sylvain Perret

1. Introduction

Sustainable production and consumption have become key policy priorities in recent years yet facing immense environmental and socio-economic challenges. Irrigation and drainage systems are instrumental to food, biofuel and fiber supply worldwide; they also have close interactions with all environmental compartments (soil, air, water) which they may affect through pollutant emissions, greenhouse gases, or excessive resource extraction. Therefore, it is crucial to assess the environmental impacts of these systems, along with an analysis of their technical and economic performances.

According to the IPCC (2006), there is no well-established approach that is able to, simultaneously and in an integrated manner, take account of the whole set of indicators that characterize the environmental, social and economic performances of agri-food systems, with regard to current global and regional sustainable development challenges. In this context, Life Cycle Analysis (LCA) has been promoted by OECD, EC, FAO, UNEP and other organizations, and is increasingly used as a comprehensive, integrated methodology for analyzing the environmental impacts of products, goods and services.

This volume has been prepared to provide insights onto the LCA methodology and its application in irrigation and drainage systems, in order to inform the ICID community and to promote LCA use.

To that aim, the ICID working group on Environment (WG-ENV) formed a task team in 2011, including Dr. SR Perret, Dr. M Van der Laan, and Prof. N Hatcho, Chair, Secretary and Vice-Chair of WG-ENV, respectively. Since 2011, the LCA task team has delivered documents and presentations, and has fostered discussions during WG-ENV meetings and ad-hoc workshops. The present document forms the main and final deliverable of the task team.

This document first presents briefly LCA principles, outcomes, and main methodological features. It proposes case studies of LCA application in irrigation and agricultural water management, as illustrations of the potential contribution of the approach.

2. What is Life Cycle Analysis ?

LCA stands for Life Cycle Analysis or Life Cycle Assessment. It proposes a systematic, quantitative and standardized methodology to assess the environmental impacts of goods, products, processes or services, in a quantitative manner.

LCA was first developed in the industry for assessing and comparing the environmental impacts of products, technological processes and options throughout their life, and for ultimately reducing the pressure onto the environment. It is now increasingly used for “eco-design”, as LCA-based research-development of alternative, more sustainable, ways of producing goods. Analysis of environmental impact towards eco-indicators and eco-labeling of products may also derive from life-cycle principles yet focusing onto some specific impact (e.g. carbon foot printing, water foot printing, energy-use labeling) (see Figure 1).

The concept of product “life cycle” means that a product is followed from its origin (“cradle”) where raw materials are extracted from natural resources through production, use, and recycling (if any) to its “grave”, the disposal (see Figure 1). The following case studies illustrate that, in fact, many LCA-related studies in the agricultural sector do not cover products from “cradle” to “grave”, but rather stop at the farm gate, and sometimes at the processing stages. The consumption, recycling and disposal stages are seldom addressed.

LCA designates at the same time the whole assessment procedure (i.e. a set of standardized methods and stages) and a set of models or algorithms that translate input and output flows into environmental impacts. In LCA, natural resource use and pollutant emissions are described in quantitative terms. Also, LCA does not always focus onto the product or service itself (e.g. irrigation technology used, or mass of rice produced), but rather onto the functions attached to it (through the concept of functional unit; e.g. volume of water delivered at crop level, or amount of calorie-equivalent produced). This allows for comparing different products or services with similar functional units.

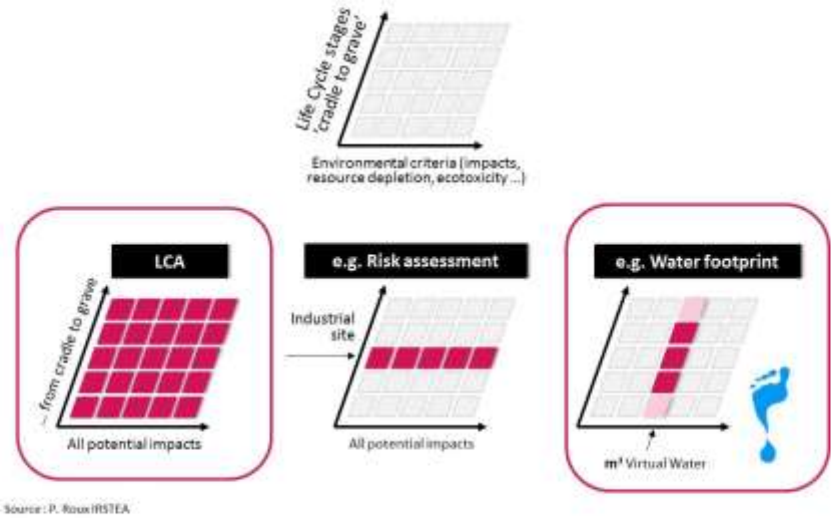

Figure 1. LCA and other environmental impact assessment approaches

As said earlier, LCA may be used for comparing products or services, for characterizing one given product (e.g. towards eco-labeling), or to investigate possible changes in a given production processes towards lower environmental impact (eco-design).

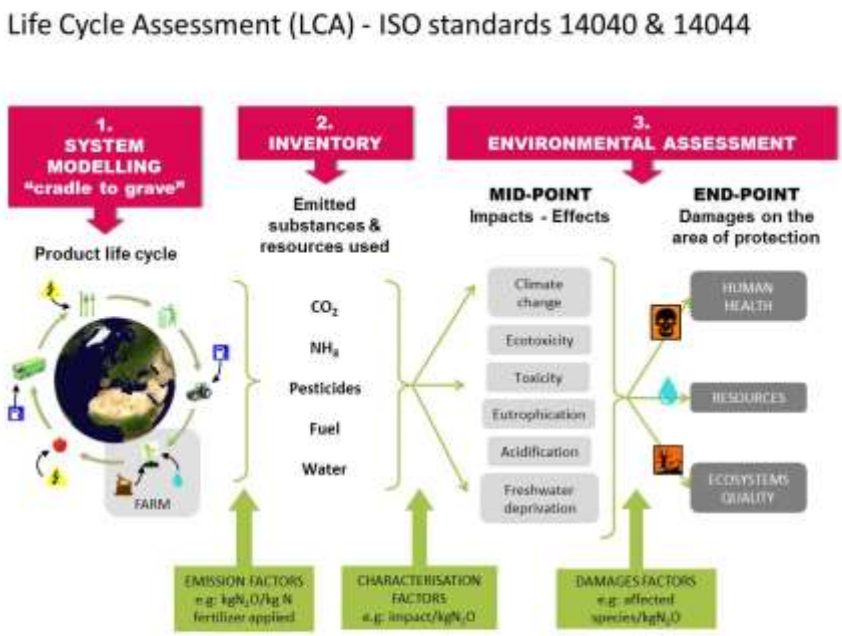

Figure 2. The main stages of LCA (source S. Payen, CIRAD)

LCA methodology is standardized (ISO 14040-41-42-43-48); as shown in Figure 2, its main stages are:

- Goal and scope definition, where the objectives and expected outcomes are defined as clearly as possible (i.e. definition of the product, good or service to study, definition of the functional unit thereof, system boundaries, allocation rules, data required, purpose and expected role of the analysis, etc.);
- Inventory analysis, where all resource uses, emissions to air, soil and water are listed, documented, quantified (i.e. a detailed input-output flow analysis), and referred to one functional unit (e.g. as per mass unit of good produced, or per unit of service rendered); this

step supposes much fieldwork, observations and primary data, in order to best reflect and document the flows at play;

- Impact analysis itself, or characterization (see Figure 2), where input and output flows are translated into environmental impacts through appropriate models and algorithms; existing databases and models are mobilized for that purpose; also specific platforms (e.g. Simapro, Gabi, etc.) are commonly used to ease up calculations and provide access to databases (e.g. Ecoinvent).

Our purpose here is not to elaborate in detail on the methodology itself but rather to introduce case studies and applications. Readers may find additional information in the following references:

An easy reader:

Baumann, H., and Tillman, A. 2004. The Hitchhiker's Guide to LCA: An orientation in Life Cycle Assessment Methodology and Application. Studentlitteratur, Lund, Sweden.

Several more methodological or case study papers on water in LCA:

Mila i Canals, L. et al., 2009. Assessing freshwater use impacts in LCA: Part I—inventory modelling and characterisation factors for the main impact pathways. International Journal Life Cycle Assessment, (42), 28-42.

Pfister, S., Koehler, A. & Hellweg , S., 2009. Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environmental Sciences and Technology, 43(11), 4098-4104.

Basset-Mens, C., Benoist, A., Bessou, C., Tran, T., Perret, S., 2010. Is LCA-based eco-labelling reasonable? The issue of tropical food productions, International conference on Life Cycle Assessment in the agri-food sector (VII), pp. 46-466.

Van der Laan, M., Jumman, A., Perret, S.R. Environmental benefits of improved water and nitrogen management in irrigation sugarcane: a combined crop modelling and life cycle assessment approach. Accepted in Irrigation and Drainage, September 2014

Payen, S., Basset-Mens, C., Perret, S.R. (2015) LCA of local and imported tomato: An energy and water trade-off. Journal of Cleaner Production, 87: 138-147

Gheewala, S.H., Silalertruksa, T., Nilsalab, P., Mungkung, R., Perret, S.R. and Chaiyawannakarn, N. (2014) Water footprint and impact of water consumption for food, feed, fuel crops production in Thailand. Water, 2014 (6): 1698-1718; doi:10.3390/w60x000x

Ullah, A., Perret, S.R. (2014) Technical- and environmental-efficiency analysis of irrigated cotton-cropping systems in Punjab, Pakistan using data envelopment analysis. Environmental Management, 54(2): 288-300

Thanawong, K., Perret, S.R., Basset-Mens, C. (2014) Ecoefficiency of paddy rice production in Northeastern Thailand: a comparison of rainfed and irrigated cropping systems. Journal of Cleaner Production, 73(2014): 204-217

It must be noted that LCA application to agricultural systems is recent, fraught with many challenges, especially in developing conditions (Basset-Mens et al., 2010): poor quality and availability of data, data scarcity, and low awareness of environmental issues. To date, applications in irrigation and drainage systems remain rare. As shown in the reference list above, there are still vivid debates about the special status of water in LCA, which require specific approaches: water is both a resource that is susceptible to be depleted and a compartment that is susceptible to be polluted. Also, the resource ultimately interacts with the three areas of protection that are defined in LCA, i.e. ecosystems, humans, and resources (see Figure 2).

3. Presentation of the case studies

The following case studies demonstrate the significant potential contribution of LCA to irrigation agriculture. They show that LCA may not be used as a single methodology but rather combined with other methodologies.

Chapter one demonstrates the potential of LCA for assessing the environmental impacts of irrigated crop production. Combining LCA with techno-economic analysis allows for an integrated approach of eco-efficiency, as a possible proxy to sustainability. Thanawong et al. combine the environmental impacts of various rice cropping systems (assessed with LCA) with techno-economic performances in Thailand, leading to defining so-called eco-efficiency indicators, as proxys to sustainability indicators. They show that systems under controlled irrigation, especially during the dry season, are more impacting and less eco-efficient than rainfed ones.

In this chapter, Box 1 was prepared by Hatcho et al. and includes an illustration of LCA use to assess the environmental impacts of environment-friendly rice systems in Japan. They show that such systems, while using fewer inputs, are not necessarily more environment-friendly than traditional ones.

Chapter two shows how LCA may be used to improve cropping systems, towards lesser impacts, with a joint use of LCA, experimentation and crop growth modeling. Van der Laan et al. combine the LCA of irrigated sugarcane with water balance and crop growth models in South Africa, in order to investigate the potential benefits of alternative, improved management of water and fertilizer. They show that improved management leads to lesser environmental impacts and sustained yields.

Chapter three demonstrates that LCA may also prove useful in infrastructure environmental assessment. Kim et al. present the results of a simplified LCA approach applied to irrigation facilities in Korea. It focuses on the impacts of two types of infrastructure, based upon water supply: a dam and a pumping station. It shows that the dam has less impact than the pumping station over a time horizon of 70 years. However, results may depend on geological conditions. Overall and as expected, the environmental impacts of the dam refer to its construction phase, while those of the pumping station refer to its operation phase.

A Life-Cycle Analysis of The Ecological-Economic Efficiency of Paddy Rice Production in Thailand

Thanawong, K., Perret, S.R., Basset-Mens, C.

Short summary

In Thailand, the rice sector is a prominent economic and policy feature, yet there are growing concerns about its sustainability. Poverty-stricken Northeastern Thailand is an essential production area for high-quality, low-yielding rice for domestic use and export. While rain-fed conditions still largely prevail, plans to extend irrigation are being drafted. This paper compares the advantages of rice production under irrigation and rain-fed conditions in both environmental and economic terms. Indicators of techno-economic performances were combined with environmental impact indicators based upon life cycle analysis, energy and water use analyses. Data were collected in 2010 at the farm level in 43 diverse rice cropping systems of Lam Sieo Yai Basin in the Northeast of Thailand, according to 3 cropping systems, namely wet-season rain-fed (Rw), wet-season irrigation (Iw) and dry-season irrigation (Id) systems. Eco-efficiency indicators were calculated as per impact category. Wide-ranging techno-economic performances and environmental impacts were observed, while cropping practices were found to be quite homogeneous. Differentiation of systems originated mostly from differences in yield, which were, in turn, mostly impacted by water supply. Yields varied from approximately 2.6 t/ha in Iw systems to 2.4 in Rw and 2.2 in Id systems. The results highlight the low performance of dry-season irrigation systems in both techno-economic and environmental terms. Id systems require mostly blue water, while the two other systems rely primarily on green water. Id systems also require more energy and labour, due to increased water management needs. Overall, the productivity of most production factors was found to be higher in Rw and Iw systems. Emissions proved relatively similar across all 3 systems, with the exception of CH_4, which was markedly lower in Rw systems due to specific water and organic residue management. Id systems systematically emitted more nitrates, phosphates and pesticides into water sources. Rw systems showed the lowest environmental impacts per ha and per kg of paddy rice produced. The Global Warming Potential was 2.97 kg CO_2-eq per kg rice in Rw systems, 4.87 in Iw systems and 5.55 in Id systems. Unsurprisingly, Rw systems were found to be more eco-efficient in most impact categories. Rw systems valued each ton of CO_2-eq emitted at approximately US$ 134, significantly higher than Iw and Id systems. This paper further discusses the results in view of contrasting perspectives, including societal objectives, farmer income and environmental integrity, and possible irrigation development in Northeastern Thailand.

1. Introduction

1.1 Rice, poverty, and the environment

Rice (Oryza sativa L.) feeds more than 3 billion people globally. Approximately 75% of the 150 million ha harvested worldwide are irrigated and provide food, income, and a diversity of ecosystem goods and services (Bouman et al., 2007a; 2007b), yet they also have negative impacts on the environment (Roger et Joulian, 1998; Tilman et al., 2001; Wenjun et al., 2006). Rice production requires large amounts of resources (water, land, energy, and chemicals), and contributes to pollution in all environmental compartments, including water and the atmosphere, due to quasi-permanently flooded (ponding) conditions. Flooded rice grows under anaerobic conditions, which favour methane formation and release. Approximately 120 g of CH_4 are released into the atmosphere for each kg of rice produced; overall, the world's rice cropping under flooded conditions contributes 13% of all anthropogenic CH_4 emissions (IPCC, 2006).

Thailand is the world's 6th largest rice producer and largest exporter. In recent years, annual paddy output has been approximately 30 Mt, with a third being exported. Rice is grown on some 10 million ha of land (or 20% of the country), with more than half grown in the Northeastern region (Isaan), the poorest region of the country. Approximately 9% of Thailand's population still lives under the poverty line; most of this population consists of subsistence-oriented, seasonal rice growers in the Isaan who sell production surplus and rely on multiple income sources for their livelihoods. Also, increasing scarcity of farm labour afflicts the region (ADB, 2012).

As a consequence, any attempt to reduce the environmental impact of rice production (through input reduction or alternative water management) or to develop irrigation should take into account the consequences with respect to economic performances such as changing yields, changing farmer income and higher labour requirements. In addition, in view of plans to extend irrigation in Isaan (Molle and Floch, 2008), there is a need to understand the comparative advantages of controlled irrigation vs. rain-fed cropping (uncontrolled irrigation during the wet season) in both environmental and economic terms.

Rice production in Isaan is currently mostly lowland rain-fed (85% of paddy land area, only in the wet season) and irrigated (15% of paddy land cover during the wet season; only 7.5% during the dry season), and shows low yields of high-quality, high-value varieties (Jasmine fragrant rice for domestic use and export). Northeastern Thailand produces approximately 80% of all jasmine fragrant rice produced nationwide (variety Hom Mali).

Rice production systems contribute 80% of freshwater extractions in Thailand, and pesticide–related toxicity is becoming a major concern. In Thailand, each ha of paddy fields requires approximately 10,000 m^3 of water per season; each kg of paddy rice produced requires 2 to 3 m^3 of irrigation water, depending on the season (Rahatwal, 2010). Significant increases in rice production through irrigation expansion in the Isaan region can only be achieved through further exploitation of the Mekong and its tributaries and wetlands, incurring the need for massive infrastructures for water diversion and potentially the destruction of natural ecosystems and harmful environmental impacts. There is currently tremendous pressure on Thailand's water resources; the country enjoys high per-capita water availability, but it ranks 14th in the world in organic water pollution and eutrophication. One third of Thailand’s surface water bodies are considered to be of poor quality; it is estimated that water pollution costs the country 1.6 to 2.6 per cent of GDP per year (World Bank, 2006). To redress these issues, Thailand has set up ambitious plans geared towards environmental protection, including climate change mitigation measures in agriculture (Office of Environmental Policy and Planning, 2000).

1.2 Eco-efficiency as a metric of sustainability

The rice-environment-poverty nexus described above relates to the sustainability of rice farms and to the possibility of reducing the environmental impact and resource use of rice cropping systems while sustaining the yields and income of farmers and the country’s position as a top producer and exporter. A workable approach to sustainability at the farm level consists of evaluating whether producers are making efficient use of resources and minimising environmental impacts while achieving their economic objectives. To that aim, economic-ecological efficiency, known as eco-efficiency (EE), may be a useful operational concept. This concept emerged in the 1990s to allow for a practical approach to sustainability (Schaltegger, 1996; Tyteca, 1996; OECD, 1998; Schaltegger and Synnestvedt, 2002; Bleischwitz, 2003). EE expresses how efficient an economic activity is with regard to its impact upon nature. EE is represented by the ratio “Product or service value / Environmental influence” (OECD, 1998). It was initially meant for the business sector to contribute to sustainable development (UN-ESCAP, 2009). The concept of eco-efficiency has been embraced by many companies and OECD countries and has proven to be a practical tool for enhancing both economic and environmental benefits. To date, it has had a focus on resource use vs. broad economic outputs (e.g., energy use vs. GDP or turn-over), and eco-efficiency has yet to fully develop at the micro level and in the agricultural sector and to consider the diversity of environmental impacts.

1.3 Approaches to economic and environmental performances

The eco-efficiency of a process is a ratio that relates the environmental loads and resources mobilised (emissions and inputs) by such a process with the economic value of the products and services provided (outputs) by the process. As such, it requires indicators of both economic and environmental performances.

Techno-economic assessment of irrigation systems and farms has long been performed. Crop budgeting, resource use analysis, productivity analysis, and farm economic assessment typically result in indicators that reflect water supply performance (Gonzales, 2000; Edkins, 2006), agricultural production performance, and the economic efficiency (productivity) of production factors such as labour, land, water, and other inputs (Ali & Taluker, 2008; Le Grusse et al., 2009; Speelman et al., 2011).

Environmental impact assessment at the same level (farm or cropping system) is much more recent. Among other methodologies, life cycle analysis (LCA), an approach to assessing potential environmental impacts, had long been identified as a potential contributor to eco-efficiency analysis (Tyteca, 1996), including in agriculture (Van der Werf and Petit, 2002). This approach is increasingly used in the industry and the agriculture sectors for assessing processes and products and for the development and implementation of environmental policies (EU, 2010a). LCA is a structured, systematic, internationally standardised method (ISO 14040 and 14044) for quantifying the emissions, resources consumed, and environmental and health impacts that are associated with the production and use of goods and services (products).

There are four main stages in LCA (ISO 14040, 2006): goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation of the results. LCA consists of a thorough and systematic inventory (life cycle inventory, LCI) of processes, emissions, resource consumptions, inputs and outputs related to the provision of a good or service. It then converts the inventory into impact indicators (midpoint or endpoint indicators, as per impact categories). This step is the life cycle impact assessment (LCIA) phase. In the LCIA phase, one may optionally apply normalisation, weighting and aggregation into single score indices (Baumann and Tillman, 2004; EU, 2010b). Finally, in the interpretation phase, the robustness of the results is discussed with regards to the quality of data used, assumptions made, and the initial goal and scope of the study. Because LCA takes into account different stages in a product's life (ideally, from the extraction of raw material, over production phases, use, recycling, to the disposal of the remaining waste), it helps track the potential shifts of environmental impacts between stages.

LCA application in agriculture has developed over the last 15 years (Audsley et al., 1997) and addressed most agricultural commodities (e.g., Williams et al., 2005). Yet, paradoxically, rice, as a crucial global commodity, has been rarely studied. To date, there is abundant literature on the assessment of greenhouse gas (GHG) emissions from irrigated paddy fields (as reviewed by Blengini and Busto, 2009). Few studies have applied LCA for assessing environmental impacts of rice production in Asia. Most published research has essentially focused on GHG and global warming potential (in Japan, Harada et al., 2007; Hokazono et al., 2009), on organic farming of rice (in Japan, Hokazono and Hayashi, 2012), and on weighting and normalization of results (in China, Wang et al., 2010). To the authors' knowledge, there are only three comprehensive published applications of LCA to rice (in Italy, Blengini and Busto, 2009; in China, Wang et al., 2010; in Japan, Hokazono and Hayashi, 2012). Basset-Mens et al. (2010) assessed the scarce rice LCA literature and highlighted the overall paucity and limitations, including a lack of consideration of the actual diversity of field and farm situations and of water and energy use. Until recently, water in LCA was only considered a qualitative compartment susceptible of being impacted upon. New methodologies on water resource depletion in LCA have been extensively investigated recently, with important breakthroughs that suggest using partial water footprinting approach (Mila i Canals et al., 2009, Pfister et al., 2009). However, empirical validation and local case studies are still lacking. Actual water consumption in agricultural systems is seldom known in developing, gravity-based conditions. Crop water requirements (CWR) and irrigation water requirements (IWR, blue water), both modelled from soil, crop and climate data, are usually used as proxies (Allen et al., 1998). The use of recent versions of FAO's CropWat (Mom, 2007; Chapagain and Hoekstra, 2011), coupled with water balance modelling in ponding conditions (Rahatwal, 2010), shows potential.

Box 1. Assessment of Environment-friendly Rice Farming in Japan Through LCA

(Authors: Hatcho, N., Matsuno,Y.,Kochi, K.and Nishishita, K.)

To reduce the negative impacts of farming, both national and local governments in Japan are promoting environmentally friendly farming. Similarly sustainable agriculture practices are pursued in different parts of the world. Shiga prefecture (135° 52′ E, 35° 00′ N), Japan is promoting such environmentally friendly farming by providing subsidies to farmers who reduce the level of chemical fertilizer application to control water pollution and eutrophication in Lake Biwa basin.

Environmental impacts of rice farming, particularly the emission of global warming gas (CO2, N2O, and CH4), eutrophication (T-N and T-P and COD to water) and energy consumption, were analyzed by applying life cycle assessment (LCA), which is a method to analyze environmental impacts associated with whole process of certain product from raw material extraction, processing/production, distribution, use, and disposal. Cultivation practices and inputs (labor, materials, and chemicals) of farmers who adopt environmentally friendly and conventional practices were collected through interviews with local farmers in the basin of Nishinoko area in Shiga prefecture. The system boundary includes all processes of paddy production from seeding to harvest/drying and machinery/materials used for production but does not include construction of facilities and buildings/land consolidation and waste disposal, distribution of products, and consumption processes. The process of making compost is also included in the analysis where compost is applied. Results show that environmentally friendly farming does not necessarily have lesser impacts when compared to conventional farming in different categories of assessment, which largely depends on the estimation of methane emission and total-P/total-nitrogen from paddy fields.

Full paper has been published by Chiang Mai University's Journal of Natural Sciences, Special Issue on Agricultural & Natural Resources (2012) 11(1): 403-408

To date, only a few research works have mobilised LCA results in eco-efficiency analysis in agriculture (in Canada: Pelletier et al., 2008; in New Zealand: Basset-Mens et al., 2009); however, these studies used modelling or scenario-based approaches and did not investigate the diversity of actual cropping systems. To the author's knowledge, no LCA-based eco-efficiency research exists in tropical agriculture under developing conditions or in rice production.

1.4 Research objectives

Given the importance of the rice sector in Thailand and growing concerns about its sustainability, environmental impacts and the embedded poverty of its farmers, this research aims at assessing the eco-efficiency of rice cropping systems in Northeastern Thailand as a main production area. In view of the currently prevailing rain-fed conditions and of existing plans to extend irrigation in Isaan, the research also compares the advantages of rice production under controlled irrigation and rain-fed conditions in both environmental and economic terms.

2. Materials and methods

2.1 Study area description

Lam Sieo Yai basin is located at the heart of the Isaan plateau in Northeastern Thailand (Figure 1) with an elevation that ranges between 100 to 200 m above sea level. Its area is 2,875 km^2. It covers 3 provinces and 7 districts, which are among the poorest of Thailand. The Sieo Yai River is the main river of Lam Sieo Yai basin. It joins the Mun river, then ultimately flows into the Mekong River. The area is exposed to a tropical savanna climate. Its average annual temperature is 18°C. As shown in tables 1 and 3, the area is exposed to two contrasted seasons: the dry season between November and April, which commonly includes severe drought conditions, and the monsoon-affected wet season between May and October, which features floods on occasions. Also, the period between December and February is significantly cooler. Annual rainfall amounts to approximately 900 mm on average yet with high inter-annual variability.

Table 1. Rainfall depth in Lam Sieo Yai basin; 30-year averages, and figures of 2010

Period		Rainfall depth (mm)
Average 30 years	Yearly	885.70
	Wet-season (July-October)	707.70
	Dry-season (February-May)	117.20
2010	Yearly	1218.93
	Wet-season (July-October)	895.98
	Dry-season (February-May)	191.55

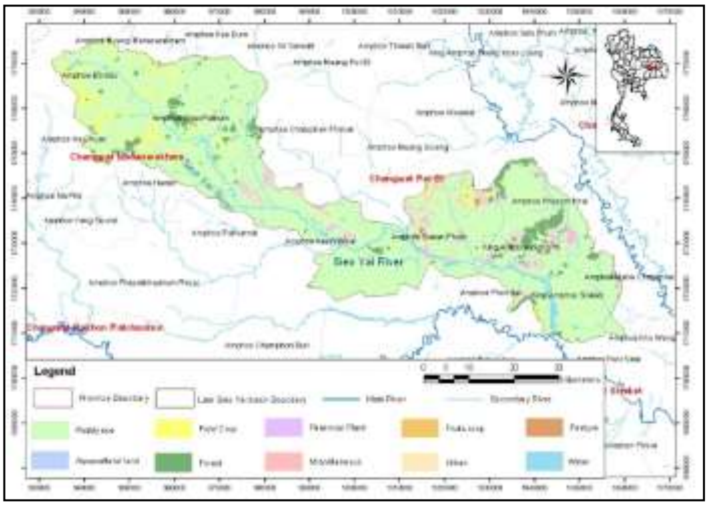

Figure 1. Map of the Lam Sieo Yai basin; location and land use (Northeast of Thailand)

In the Lam Sieo Yai basin, 83% of the total area is agricultural land, of which 96% is covered with paddy fields. In the basin, 75% of paddy fields fall under the Sieo Yai Irrigation Project and benefit from controlled water supply. The other 25% are rain-fed paddy fields of individual farmers. Lowland rain-fed rice is grown only during the wet season, while irrigated rice is cultivated during both seasons. Rain-fed conditions refer to conditions of lowland rice that is cropped under flooding conditions with no control of water supply. Rainfall, soil moisture, and natural runoff alone (green water) provide water to the paddy fields. Figure 2 shows a simplified sketch of water flows in a paddy field. In Figure 2, the outflow (drainage) is hardly happening because farmers let the water evapo-transpirate and percolate well before the end of the cycle, and usually do not have to pump water off the fields.

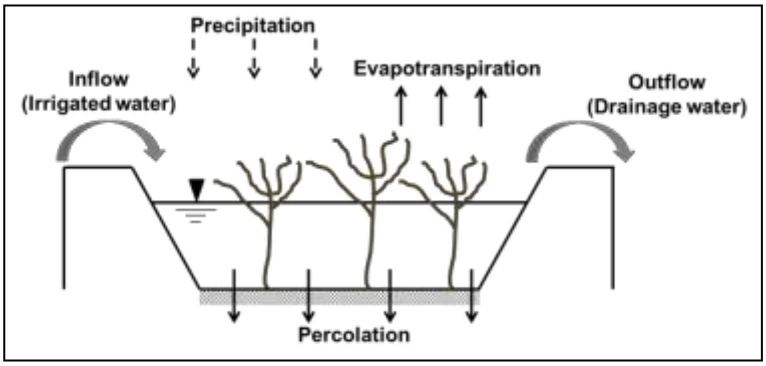

Figure 2. A representation of water flows in a paddy field

2.2 Joint LCA and techno-economic analyses

2.2.1 General approach

The research collected, analysed and combined indicators of techno-economic performances (rice production, costs, and product value) with environmental impact indicators based upon the life cycle approach. Both approaches apply at the same plot level (cropping system level) and complement each other. Techno-economic analysis typically results in monetary values as per factor of production (e.g. labour, land, agro-chemicals) while LCA expresses environmental impacts as per selected functional units (in this case: mass of product and area of land used). The research reported here is problem-oriented; it focuses on midpoint indicators for different environmental impact categories (e.g., global warming potential, eutrophication, or acidification) and resource use (land, water and energy). Overall, the chosen approach is of an accounting nature (as opposed to a change orientation, which would require technological scenarios). The performed LCA is therefore attributional and static. The primary functional unit (FU) for LCA is the mass (kg) of raw paddy rice (unmilled) at the farm gate (approximately 15% h.c.). The secondary FU is 1 ha of land used. A third FU "hidden" is 1 dollar of profit earned, because eco-efficiency is a ratio that expresses how many dollars are made as per impact, which is the reverse ratio of impact as per dollar made, as expressed in LCA. Total value product (or gross income, i.e. market price of product multiplied by mass of product) has been used to represent the total economic value of the product.

All data were collected, calculated or modelled in diverse typical rice farming situations of the Lam Sieo Yai basin in Northeastern Thailand. LCA and economic results were finally used to calculate eco-efficiency indicators as per impact category.

Table 2. Average monthly rainfall (mm) in Lam Sieo Yai Basin (30-year average)

Month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Monthly rainfall (mm)	10	2.3	18.5	16.4	80	43.4	142.1	202.9	259.7	103	5.4	2

2.2.2 Systems, and systems' boundaries

Three cropping systems were investigated based upon water management system: wet-season rain-fed rice (Rw), wet-season irrigated rice (Iw) and dry-season irrigated rice (Id). Although the traditional transplanting of sprouts from nursery to paddy field may still be observed, the direct sowing of dry seeds has recently become overwhelmingly predominant in Northeastern Thailand. Seventy-five per cent of farms have adopted the technology of dry-seed sowing, which spares time and labour but results in lower yields. The results presented here refer to this planting mode, which was carried out in each water management system. Two fragrant rice varieties are chiefly cultivated in Northeastern Thailand: Kao Dok Mali 105 (during the wet season) and RD15 (during the dry season).

Primary data were collected by means of field observations and interviews with farmers; data refer to the two cropping seasons of 2010, including dry and wet seasons. Table 1 shows the precipitation conditions that prevailed during these seasons compared to long-term averages; it highlights the fact that 2010 received more precipitation than 30-year averages, on both a yearly basis and a per-season basis. Fifteen farm plots were selected and studied for wet-season rain-fed and irrigated rice systems and 13 farm plots for dry-season irrigated rice system. Both environmental impact analysis and techno-economic analysis were performed on all 43 cropping systems.

We decided to report most results as median values, with minimum and maximum values. The reasons for this decision are manifold: most underlying biophysical processes leading to agricultural performances and environmental impacts are not linear; the calculations leading to the assessment of direct field emissions and environmental impacts are not of a linear nature either as a result of threshold effects due to discrete scaling factors related to crop and water management practices; and consequently, the results do not follow normal distributions (e.g. in figures 4-5-6).

Although LCA conceptually covers the whole life cycle of a product or service, the present study covered the rice production systems from “cradle” (mobilisation of all raw resources and equipment) to farm-gate (unmilled rice); we did not consider further rice processing, storage, transport, packaging, consumption, or other aspects (as shown in Figure 3). This choice was justified by the fact that approximately 60 to 90% of global warming impact of rice relates to production at field level (Harada et al., 2007; Hokazono et al., 2009); furthermore, Blengini and Busto (2009) found that most other environmental impacts are predominantly generated at the farm level. The flow diagram of the studied systems is shown in Figure 3, which describes the sequence of typical operations in rice cropping systems of Northeastern Thailand. In Figure 3, the flows related to machinery and equipment include those resulting from manufacturing, transportation and direct use (fuel consumption). Flows related to seeds and chemicals refer to flows resulting from production and transportation. Human labour is being considered only in techno-economic calculations.

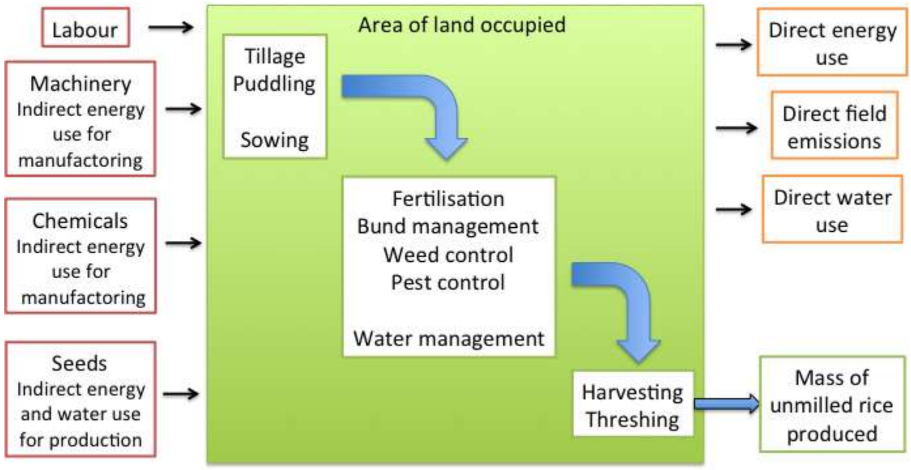

Figure 3. Flow diagram for the studied rice cropping systems

2.2.3 Joint LCA and techno-economic inventories

The common technical data and specific data needed for LCI and economic analyses for the main stages of rice production (land preparation and sowing, rice cultivation and field operations, harvesting) are presented in Table 3.

Table 3. Type and source of data needed for LCI and economic analyses (Note: only technical data are common to both LCI and economic analysis)

	Areas of inventory	Data sources	Unit
Technical data	Input use (seeds, chemicals) Direct energy consumption (machinery, porTable equipment)	Primary data (farm level) Primary data (farm level)	Kg or g MJ
	Water consumption	Modeling from IWR (Water balance model and CropWat)	m^3
	Yield Land use	Primary data (farm level) Primary data (farm level)	kg/ha ha/production cycle
LCI	Indirect energy consumption (from manufacturing and transport of machinery and chemicals) Direct field emissions	Ecoinvent database (in SimaPro) Modelling (secondary data IPCC and tier-2 references)	MJ Kg substance per
Economic data	Production costs (labour, chemicals, machinery, energy) Economic value (total value product) Labour	Primary data (farm level) Primary data (market price at farm gate) Primary data (farm level)	Thai Baht Thai Baht h

2.3.1 Inventory of field operations and performances

The inventory data required to perform both techno-economic analysis and environmental impact assessment comprise the following processes and operations:

- Field operations with machinery (ploughing, puddling-rolling, combine harvesting); data collected include type, weight, scheduling, use time, use costs, and labour requirements,
- Field operations performed manually (sowing, fertiliser application, bund maintenance, water management, spraying); pesticide-spraying is performed manually with porTable equipment; water management at the plot level requires porTable water pumps; bund maintenance involves grass-cutting with porTable equipment; data collected include type (chemical and equipment if any), capital value, scheduling, use time, use costs, and labour requirements,
- Inputs and agro-chemical use (seeds, urea, 15-15-15, 16-16-8, glyphosate, $CaCO_3$, isoprocarb, metaldehyde); data collected include type (commercial name, brand), cost, doses, scheduling of application, labour requirements,
- Yields and market price at the farm gate,
- Cultivation and harvested areas.

These data were collected through detailed questionnaires and farmer interviews at the farm level (related to a given plot under study) during the 2010 cropping seasons.

Water use through irrigation was modelled with CropWat (FAO, 1992) and water balance. The inventory for the manufacturing and delivery of machinery and agrochemicals equipment, machinery, inputs and energy carriers used during field operations were calculated with SimaPro 7 from field data and based upon existing conversion rates, methods, and databases (Ecoinvent database).

2.3.2 Direct field emissions

The following emissions to air were considered: CH_4, N_2O, NOx, and NH_3. These emissions were modelled based upon the norms established by the International Panel on Climate Change (IPCC, 2006), adjusted with secondary, region-relevant information (Yan et al., 2003a; 2003b). Carbon dioxide was considered neutral (Williams et al., 2005).

It must be noted that, while the IPCC (2006) suggests a default average baseline emission (EF_c) of 1.30 kg-CH_4.ha^{-1}.d^{-1} (with high variation), Yan et al. (2003a) recommend EF_c = 3.12 kg-CH_4.ha^{-1}.d^{-1} as the baseline emission factor based upon direct field measurements in Northeastern Thailand, where specific conditions prevail (high soil, air and water temperatures and high solar radiation, which have been shown to be determining factors of increased CH_4 emissions). All scaling factors affecting EFc were taken from IPCC (2006) according to observed local crop and water management practices: rain-fed conditions (uncontrolled, intermittent flooding with multiple aeration phases; non-flooded pre-season of more than 180 days; straw incorporated more than 30 days before cultivation), and irrigation (with multiple aeration phases; non-flooded pre-season of less than 180 days; straw incorporated less than

30 days before cultivation). With regards to common practices in the study areas, organic amendments include only rice straw and rooting systems that remain after harvesting. The literature commonly considers a dry grain / dry straw ratio of 1:1. According to average grain yields in recent years in the study area, it was assumed that 2.7 tons of dry straw were incorporated per ha as organic fertiliser.

According to these conditions and relative scaling factors, the application of the IPCC's CH_4 emission model results in adjusted daily emission factors of 1.552 kg-CH_4.ha^{-1}.d^{-1} in rain-fed conditions and 3.511 kg-CH_4.ha^{-1}.d^{-1} under irrigation conditions (for both dry and wet seasons). The observed average length of cropping cycles is 120 days, from sowing to harvesting. It is very homogenous, although dictated by rice ecophysiology and climatic conditions, and also by the availability of harvesting equipment, which is rented to local entrepreneurs (combine harvesters).

Because flooded conditions are unfavourable to nitrification, N_2O and NO_x emissions to air have long been assumed to be negligible in paddy rice production. Yan et al. (2003b) reviewed literature with measurements of N_2O and NO_x emissions from continuously flooded paddy fields and proposed emission models that included both baseline and fertiliser-dependent emissions and were specific to paddy rice produced in South Asia but not Thailand. These models were adjusted to the lengths of cropping seasons in each sampled case (average: 120 days); however, these models failed to consider intermittent flooding conditions with drying periods during which more active nitrification-denitrification occurs, most likely leading to higher N_2O and NO_x emissions.

Yan et al. (2003b) focused their literature analysis on urea-induced NH_3 emissions because urea is the most common chemical fertiliser used by farmers in South and South East Asia (urea and ammonium-based fertilisers form approximately 85% of all nitrogen fertilisers applied to paddy fields in Northeastern Thailand). They proposed a model of urea-induced NH_3 emissions that depends upon the timing and mode of application, which, in turn, have a strong influence on the volatilisation rate. In spite of a paucity of data, the same authors also proposed NH_3 emission factors for other nitrogen-based fertilisers. These models were used, with adjustment to a 120-day cropping season.

Water-soluble nitrates and phosphates have been considered to be the two potential pollutants emitted to the water compartments during rice cropping. A similar approach was carried out for both of these pollutants. Paddy rice consumes significantly more ammonia than nitrates, in contrast to other global crops. Because urea and ammonium-based fertilisers prevail in Northeastern Thailand, direct nitrate emissions result mostly from biochemical transformations (e.g., denitrification) and the whole nitrogen cycle and balance rather than from direct fertiliser loss. The principles underlying the nitrate emission assessment are that (1) nitrates form the remaining components of the overall nitrogen mass balance, the other components of which were determined in earlier sections; (2) these water-soluble nitrates may leach to the water compartment through surface drainage and deep percolation; and (3) such a portion refers to the ratio between water that is not used by the crop and overall water supply; in other words, it relates to water use efficiency.

Accordingly, nitrates potentially leaching from a paddy field are modelled according to a dual N and water mass balance approach suggested by Pathak et al. (2004). N inputs include fertiliser, precipitation, irrigation water and soil (N stock, immobilisation). N outputs include losses in surface runoff, deep percolation, harvested and exported crop components (mostly rice grain), soil losses (erosion), mineralisation, volatilisation and denitrification processes. The difference in N stored in pre-cultivation soil and in post-cultivation soil is considered negligible because these soils have maintained long-term sTable nitrogen contents under the same cropping systems for years. Similarly, the organic matter dynamic is deemed balanced over time, with equal mineralisation and immobilisation (straw). Other components, such as biological nitrogen fixation, groundwater contribution, and exports by weeds, are ignored (Pathak et al., 2004).

All components of N balance therefore are known, assumed or neglected, with the exception of N losses to deep percolation and surface drainage as water-soluble nitrates. N inputs from fertiliser have been calculated from the fertilisers' formulae and application doses. N inputs from rainfall and irrigation water were calculated from data on N contents, average precipitation and irrigation data over the period under consideration (cropping cycle). The Pollution Control Department of Thailand suggests an average NO_3 concentration of 0.7 mg.l^{-1} in precipitation, and 0.11 mg.l^{-1} in irrigation water. Rainfall data from the Thailand Meteorological Department rainfall stations located in the study area were used (as shown in Table 2).

N uptake from rice plants was calculated from the average N contents of the average mass of exported crop parts (grain and ears). N losses due to emissions to the air in the form of N_2O, NO and NH_3 were calculated as shown in previous sections. N_2 is emitted during the last phases of denitrification. Although not a pollutant, N_2 needs to be assessed to complete the whole mass balance. Brentrup et al. (2000) proposed an emission factor of 9% of all N fertilisation. Although their emission factor corresponded to annual crop conditions under temperate climate, it was used in this study, in the absence of more adapted data.

It was assumed that the remaining components of nitrogen mass balance were nitrates. Water-soluble nitrates may be either absorbed by the crop through evapotranspiration flux or emitted to the water compartment as pollutants via deep percolation and drainage. It was also assumed that the proportion of nitrates bound to drain or leach to the surface and groundwater compartments during the crop cycle equalled the proportion of water that was unused by crops in the paddy system. Therefore, a water mass balance was needed to ascertain water use efficiency and to determine percolation and drainage components. Runoff was considered nil because in common conditions, paddy fields are flat and managed in a way that prevents water from spilling over bunds; farmers maintain water depth between defined minimal and maximal ponding conditions (generally 0 to 150 mm). However, at times, and especially at the end of the cropping season, farmers drain the fields off.

Average monthly rainfall data (as shown in tables 1-3) and reference evapotranspiration data provided by meteorological services were used, as well as typical irrigation data collected in the study area. Crop coefficients (Kc) are required to assess actual evapotranspiration and were drawn from FAO and from local references by the Royal Irrigation Department of Thailand. The CropWat platform (FAO, 1992) was used to calculate actual evapotranspiration.

A similar approach was applied to phosphates, under similar assumptions regarding the stability of long-term contents, the absence of erosion, and with similar modelling approach (water mass balance). P inputs from fertiliser were calculated from fertiliser formulae and application doses. According to the Pollution Control Department of Thailand, the average value of P concentration in precipitation in Thailand is 0.045 $mg.l^{-1}$; the average P concentration in irrigation water in Thailand is 0.125 $mg.l^{-1}$.

In the cropping systems under study, the pesticides typically used include a molluscicide (solid pellets, metaldehyde-based), an insecticide (liquid, isoprocarb-based with $CaCO_3$ as humectant additive) and an herbicide (liquid, glyphosate-based,); all are hand-sprayed at different stages while the field is flooded most of the time. It was assumed that 100% of pesticides ultimately end up in both soil and water compartments because none of the pesticide is supposed to concentrate in the rice grain and leave the field at harvest. Straw and rooting systems are left in the field to decay. Under these circumstances, it was arbitrarily decided to split emissions equally between soil and water compartments (50%-50%).

2.4 LC impact assessment and eco-calculations

Impact assessment is the third stage of LCA. Because there is still no consensus on weighting, impact assessment was focused on characterisation, as suggested by Blengini and Busto (2009). The selected indicators include resource-use indicators: energy use (EU), freshwater use (WU) and land use (LU); they also include environmental impact (mid-point) indicators: eutrophication (EP), acidification (AP), global warming potential (GWP_{100}), freshwater aquatic ecotoxicity (FWAE), ozone depletion (ODP). These impact categories were chosen based upon their widespread use in agricultural LCA studies, allowing for comparison. More specifically, FWAE was selected because freshwater is a key feature and compartment of paddy rice cropping systems. Characterisation was performed with the SimaPro platform using CML baseline 2000/world, 1995 methodology.

The GWP for a 100-year time horizon (GWP_{100}) was calculated according to IPCC in kg CO_2-eq. (Guinée et al., 2002). With factors recommended by Guinée et al. (2002), EP was calculated in kg PO_4-eq, FWAE was calculated in kg 1-4 dichlorobenzene (DB) eq, and ODP was calculated in mg trichlorofluoromethane (CFC-11) eq. AP was calculated using the generic method proposed by Heijungs et al. (1992) in kg SO_2-eq. Energy use refers to the depletion of energetic resources and was calculated based upon direct and indirect fossil fuel use, including physical (machinery) and chemical (fertilisers and pesticides) energy; all were converted to MJ. Water use refers to the volumetric depletion of water resources and was calculated based upon water footprint concepts. Crop evaporative consumption was modelled with water balance and CropWat models (FAO, 1992); it included the evaporation of rainfall from crop land (green water use, WU_g) and the evaporation of irrigation water from crop land (blue water use, WU_b). Land use refers to the loss of land as a resource in the sense of being temporarily unavailable for other purposes. Details on CML 2002 calculations, impact factors and normalisation may be found in CML (2002). CML 2002 methodologies and necessary databases are included in the SimaPro 7.3 modelling platform (Pré Consultants, 2010a; 2010b), which was used for this research. Commercial pesticides were modelled according to their active ingredients and the inventory data from Ecoinvent database within SimaPro.

The eco-efficiency of the rice cropping systems was quantified by expressing the total value generated (gross income or Total Value Product) as per environmental impact created (for each impact category). Net return as per environmental impact was also calculated (net income, or gross income minus total production costs) to represent eco-efficiency from the farmers' perspective.

3. Results[1]

3.1 Utilisation of production factors and performances per area cultivated

Table 4a shows the techno-economic performances of the three cropping systems per area cultivated (ha). The results highlight the low performances of dry-season irrigated rice systems (Id), the production factor requirements of which are systematically higher than those of the two other systems; in addition, the Id system yielded markedly lower production. This system also requires mostly blue water (irrigation water), while the other two rely predominantly on green water (natural stocks and flows). The Id system requires 3 pumping episodes on average to replenish ponding conditions in paddy fields; therefore, it requires more labour and energy (pumps).

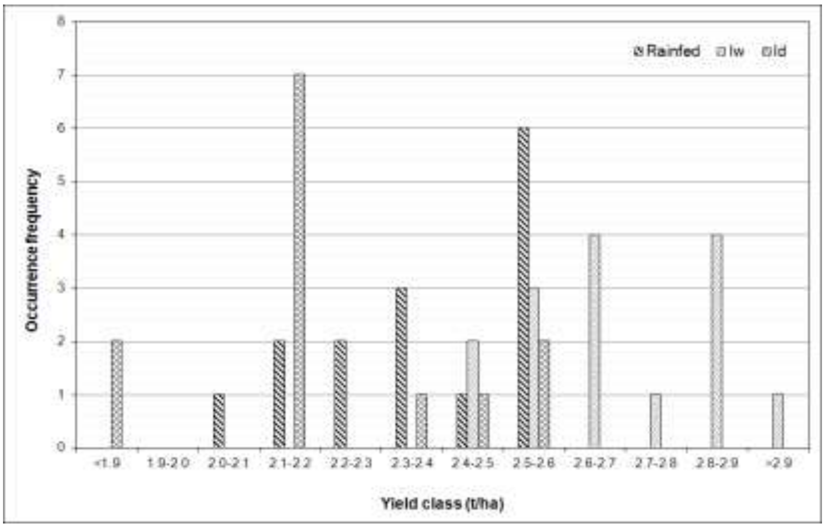

Figure 4. Paddy rice yields recorded among the 43 rice cropping systems in 2010 (Rain-fed, Iw: Irrigated wet season; Id: Irrigated dry season)

Labour, energy and pesticide requirements are markedly lower in rain-fed conditions due to lesser water management requirements (no pumping) and an absence of treatment against the golden snail (Pomacea canaliculata) which cannot reproduce during the cropless dry season of rain-fed plots. Energy requirements are consistent with the values reported by Pimentel (1980) and consist of approximately 12,000 MJ/ha and 15,000 MJ/ha for rice production in the Philippines in the wet and dry season, respectively (excluding human power).

The high level of homogeneity of fertiliser and pesticide application practices within each cropping system resulted in relatively homogeneous production costs per system; however, there were diverse outcomes in terms of yield (as shown in Figure 4 for the 43 cropping systems -year 2010) and therefore of gross and net income. Net income per system was wide-ranging, with the Id system being the least profiTable and the most variable. Conditions during the dry season are less favourable temperature-wise and more uncertain and variable in terms of water management. Iw systems showed higher homogeneity of results and a potential for the highest yields and net income.

3.2 Productivity of production factors and performances per mass of rice produced

Table 4b shows the productivities of production factors and the techno-economic performances of the three rice cropping systems. Overall, the results confirm that the productivities of most factors are higher in the Rw system, in which farmers produce more rice per labour unit, pesticide unit and total energy unit. Interestingly, the productivities in the Rw and Iw systems are similar for factors such as fertiliser,

[1] Tables 4a, 4b, 5, 6a, 6b, 8a, 8b feature at the end of the chapter, as appendices.

total water and green water. Return on investment (mass of rice produced per production cost) is slightly higher in the Iw system compared to the Rw system (0.117 kg/THB and 0.114 kg/THB, respectively) and is lowest in the Id system (0.095 kg/THB). Median yields (land productivity) vary from 2,625 kg/ha in the Iw system to 2,375 in the Rw system and 2,188 in the Id system. Finally, the amount of rice per net income unit is markedly lower in the Iw system (0.297 kg per THB earned as net income) and Rw system (0.310) compared to the Id system, in which farmers need to produce twice as much rice (0.662 kg) to obtain the same net income.

3.3 Direct field emissions and environmental impacts

Table 5 reports the direct field emissions that were calculated. Emissions to air proved relatively homogeneous across all three systems, with the noTable exception of methane emissions. Rw systems emit a median amount of 76 g CH_4 per kg of paddy rice, compared with 158 g and 176 g for Iw and Id systems, respectively. Lower CH_4 emissions in rain-fed conditions relate first to the water regime in the pre-season before the cultivation period (non-flooded conditions for more than 180 days) and second to the management of organic residues (incorporated more than 30 days before cultivation). CH_4 emission figures broadly concur with those of the IPCC (2006), which reports that approximately 120 g of CH_4 are released into the atmosphere for each kg of rice produced; however, our results reveal significant local differences based on cropping systems and water management practices. With regards to emissions to water, Id systems systematically emit more nitrates, phosphates, and agro-chemicals per both functional unit, on account of the overall lower productivity of chemical inputs.

Tables 6a and 6b report the environmental impacts for selected impact categories, per ha occupied for cultivation and per kg of unmilled rice produced, respectively. Overall, LCIA confirms the results related to direct field emissions and resource-related results of the techno-economic analysis. On a land use basis (Table 6a), GWP_{100} is markedly different between rain-fed and irrigated systems, Iw showing the highest impact. Differences in CH_4 emissions were previously discussed (straw incorporation and water management during pre-cultivation times) and explain this result. In all other impact categories, Rw systems systematically show lower impacts per ha than Iw and Id systems, with the latter having the highest impacts. However, AP, ODP and total water use are of the same magnitude across systems.

When impacts are expressed per mass of paddy rice produced (Table 6b), the impacts of Id systems are even higher than those of the two other systems due to the lower yields. GWP_{100} becomes higher in Id systems (5.55 kg CO_2-eq) compared to Iw systems (4.87). Rw systems remain the least impacting with 2.97 kg CO_2-eq. Figure 5 shows the diversity of GWP_{100} results obtained from calculations on all 43 sampled cropping systems. Although wide-ranging, the results clearly differentiate the three cropping systems. Total energy use is higher in Id systems (9.635 MJ per kg rice) compared to Iw and Rw systems (7.5 and 7.285, respectively).

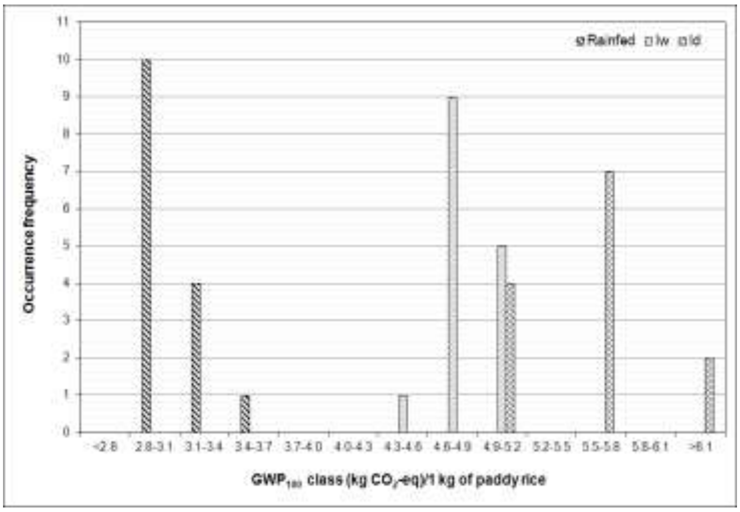

Figure 5. Global warming potential over 100 years (GWP_{100}) per kg of paddy rice produced calculated for the 43 rice cropping systems (wet and dry seasons 2010) (Rain-fed, Iw: Irrigated wet season; Id: Irrigated dry season)

Figure 6 shows the diversity of water consumption in the sampled cropping systems. Variations in water use are especially marked in dry-season irrigation, showing diversity of practices in farmers' decisions and strategies regarding water supplies (pumping episodes).

Table 7 reports a contribution analysis on rain-fed paddy rice, showing the relative contribution of cropping subsystems to each impact category. Direct field emissions to air and water are likely to overwhelmingly contribute to AP, EP, GWP_{100} and FWAE. Field operations, meaning operations requiring the use of machinery and equipment (including water pumping, and the manufacturing of all equipment) contribute 20% of all energy use and a large part of ODP.

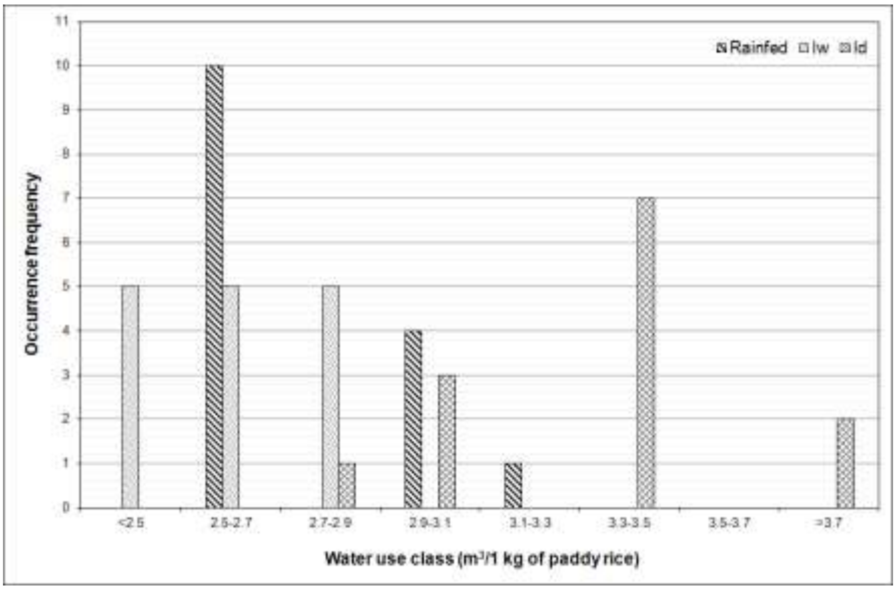

Figure 6. Total water use (WU) per kg of paddy rice produced modelled for the 43 rice cropping systems (wet and dry seasons 2010) (Rain-fed, Iw: Irrigated wet season; Id: Irrigated dry season)

Fertiliser application and manufacturing contribute a majority of total energy use, a large part of ODP, FWUE, and a marginal amount to AP, EP and GWP_{100}. Pesticide application and manufacturing contributes marginally to total energy use. Rice seeds also contribute marginally to FWAE and EU. Pesticide application requires small amounts of water, and the main contributor to WU remains crop water use. Overall, direct field emissions are contributing a main part of input-related impact categories at local and regional scales (AP, EP, FWAE) and on the global scale (GWP_{100}); they mostly depend on water management practices for methane emissions, and both agro-chemical and water management for other emissions. As stated by Blengini and Busto (2009), this predominant role calls for more reliable and site-specific data. Contribution analysis of the two other irrigated systems shows the same structure and overall contributions, although total water use in Id systems results mostly from blue water use (irrigation water), while WU in Iw systems results mostly from green water use (natural stocks and flows).

Table 7. Contribution of sub-systems to the impacts of rain-fed paddy rice production

Wet-season rain-fed rice							
Subsystems	Percentage of impact to subsystems						
	AP	EP	GWP100	ODP	FWAE	WU	EU
Direct field emissions	82.70	96.36	61.88	0.00	57.94	0.00	0.00
Field operations	12.12	1.76	27.29	48.16	1.90	0.00	20.00
Fertilisers (manufacturing and transport)	4.64	1.71	8.68	46.17	31.52	0.00	66.67
Pesticides (manufacturing and transport)	0.00	0.00	0.02	0.43	0.31	0.003	4.44
Rice seed production	0.54	0.16	2.13	5.23	8.32	4.999	8.89
Crop water use	0.00	0.00	0.00	0.00	0.00	94.997	0.00

3.4 Eco-Efficiency and net return to environmental impact

Table 8a reports the eco-efficiency of the three systems as per impact category. Because market price (the market value) of paddy rice is assumed to be identical in all three systems (12 THB per kg), the results are basically reversed values of the results on impact per kg of rice produced shown in Table 6b. However, there is an interest in reporting eco-efficiency as such, as it represents how cropping systems generate total value per environmental impact unit they create. In that sense, Rw systems are more eco-efficient than others, with the exception of AP, ODP and LU impacts, for which Iw systems perform slightly better. Id systems lag significantly behind the other two systems. Interestingly, Rw systems value each ton of CO_2-eq emitted at 4040 THB, or approximately 134 US$ per ton. Iw and Id systems value each ton of CO_2-eq emitted at 82 and 72 US$, respectively.

Table 8b reports the net return on environmental impact, that is, the net income left to farmers per environmental impact unit. It represents how cropping systems generate income for the farmers per environmental impact they create. The results show that Iw systems are more “net return efficient” than others, with the noTable exception of GWP_{100} for which Rw still performs better. Id systems still lag far behind the other systems in terms of net return efficiency.

4. Discussion

4.1 Homogeneity of practices, diversity of performances and impacts

Farmers’ practices proved surprisingly homogenous across cropping systems, showing particularly small variations in water use, and application of agrochemicals. Production costs per ha illustrate such relative homogeneity of practices. The limited sample size may hide the actual diversity; also, farmers may have responded to questionnaire-based interviews in a generic way, focusing on recommendations they receive rather than on their actual varying practices. Indeed, in Thailand’s irrigation projects, technical support is provided by local officers of the Royal Irrigation Department (RID) that manages the projects, in association with agro-chemical retailers; all tend to promote and disseminate blanket recommendations. Further, collective water management in irrigation systems imposes synchronicity and commonality of practice, in single-crop systems where both rice physiology and climatic conditions prevail over individual contingencies and liberty. The homogeneity of practices is less comprehendible with regards to rain-fed cropping systems, performed by individual farmers, least connected to RID. Small-scale paddy farmers often lack the education and own experience to challenge existing norms and to experiment. Thailand rice farmers are generally very abiding of norms and standards set up by authorities. Strikingly, labour use shows much more diversity, although it is also dependant on water management. Labour mobilisation in a cropping system typically refers to one individual farmer’s decision and organisation mode; contingencies and strategic choices can more fully materialise.

In spite of the relative homogeneity of cropping practices, overall and per sub-cropping system, outcomes in both economic and environmental terms show significant diversity. Net income and global warming potential are particularly wide-ranging in the different systems. This variation mostly results from large differences in yields, overall and per sub-cropping system. Yields and resulting net incomes are more diverse (less stable) in Rw and Id systems compared to Iw systems, due to a lack of control of the water supply and a lack of water, respectively. Attempts to relate farmers’ performances to several socio-economic factors at the household level (i.e., experience in farming, age, level of education) proved unsuccessful. Instead, it was observed that, while Id farmers usually try to refill their paddy fields three times per season, many do not actually obtain enough water (e.g., canal tail-enders). The precipitation levels of the dry season of 2010 were relatively high compared to 30-year average precipitation levels; the lack of water for Id system farmers could have been even more damaging to yields in normal or drier years. This would potentially result in lower yields, and increased differences in performances and impacts between wet season and dry season systems. The same reasoning applies to Rw systems, which showed relatively high performances and low impacts in 2010 but would perform well below Iw systems under drier conditions.

4.2 Environmental impacts: convergences and discrepancies with other studies

Three published studies of rice from Italy (Blengini and Busto, 2009), China (Wang et al., 2010) and Japan (Hokazono and Hayashi, 2012) were chosen for the comparison with our study for North East Thai rice. All three studies, although showing contrasted goal and scope, had enough transparency in materials and methods to allow for calculating LCA results per kg of rice at-the-farm-gate and in the same units. Interestingly, none of the available studies presented toxicity results and used a reduced selection of impacts categories (4-6).

For water use, our results (2.646-3.317 m^3/kg rice) were in the range showed in other studies, from 0.431 m^3 for Wang et al. (2010) to 4.9 m^3 for Blengini and Busto (2009). However, apart from WU, our results for Thai rice were either of similar magnitude yet greater (energy use, GWP, ODP), or much greater (Acidification and Eutrophication potentials) compared to the results from other regions. This trend of LCA results per kg of rice being greater in our case study can globally be explained by rice yields being markedly lower in the Isaan region of Thailand as well reflected by the sampled systems. While yields can reach easily 4 to 6 tons per ha, and even more, in the Central Plains of Thailand and in neighbouring countries, they hardly reach 2.5 tons in Isaan, due to the specific, high-quality, high-value, low-yielding varieties of fragrant rice used (Hom Mali). As showed previously, GWP_{100} per kg of rice in our case study ranged between 2.97 and 5.55 kg CO2-eq against a range between 1.46 kg CO2-eq (Hokazono and Hayashi, 2012) and 2.374 (Blengini and Busto, 2009) from the literature. In addition to the lower yields, the greater GWP result can be further explained by the use of the CH_4 baseline emission value suggested by Yan et al. (2003a) that is higher than the generic one suggested by IPCC (2006) for paddy rice, on account of specific pedoclimatic conditions in Isaan. Our results on energy use (7.3 – 9.6 MJ per kg of rice) and ODP (0.068-0.082 mg CFC11-eq per kg of rice) were similar to those obtained by Blengini and Busto (2009) on Italian rice in highly mechanised field conditions (8.75 MJ for non-renewable energy use and 0.06 mg CFC11-eq for ODP). Conversely, our results for AP (0.040-0.049 kg SO2-eq) and EP (0.075-0.099 kg PO4-eq) were much greater than the values found in the literature ranging for AP from 0.00616 kg SO2-eq for Blengini and Busto (2009) to 0.024 kg SO2-eq for Wang et al. (2010) and for EP from 0.00678 kg PO4-eq for Blengini and Busto (2009) to 0.013 kg PO4-eq for Wang et al. (2010). These impact categories are mostly affected by field emissions of NH_3, NO_3 to water and P to water. As for CH4 emissions, specific emissions factors or equations were used to estimate field emissions in our case study using equations from Yan et al. (2003b) for estimating ammonia emissions and a combination of nutrient budgets (N or P) and a precise water balance for the studied systems for N and P to water. The greater AP and EP in our study might therefore reflect more favourable conditions (e.g. higher temperatures) for these emissions compared to other situations. However, the insufficient level of detail and transparency in published LCA studies makes also possible certain discrepancies in the methods used across studies. Harmonised methods and assumptions would be desirable to complete LCA study comparisons across contrasted situations.

4.3 Sustainability and the comparative advantages of rain-fed rice cropping

The results contribute insights and data to the debate on the need to further develop irrigation in the context of North-eastern Thailand, with necessary precautions due to limited data. Rain-fed systems are reasonable alternatives and compete well against irrigation during the wet season. Proponents of irrigation development in North-east Thailand advocate that rain-fed systems only provide cropping opportunity during the wet season and force farmers to resort to alternative livelihoods in the dry season. In any case, the Isaan region has a long tradition of rural seasonal outmigration during the dry season and of off-farm and on-farm diversification of livelihood systems. It seems that irrigation during the dry season is not very profiTable or environmentally friendly; in addition, this cropping system requires significant amounts of blue water, which must be tapped from existing limited resources at the expense of other users or the environment.

For a societal objective of higher rice production and limitation of outmigration, irrigation during both seasons guarantees higher production overall, and keeps farmers busy all year round.

From a farmer’s viewpoint, dry-season irrigation requires more inputs, higher costs and labour, and ultimately shows lower efficiency. Furthermore, if eco-efficiency and environmental integrity are factored into decisions, irrigation during dry season is clearly not the best option.

Further, the striking shift from traditional transplanting to direct sowing of dry seeds illustrates the fact that rice farmers in Isaan are seeking labour efficiency and time-saving solutions, rather than high yields, in a context of labour scarcity, massive seasonal outmigration, and diversified rural livelihood systems (ADB, 2012). Indeed, direct seedling results in lower yields than transplanting, yet with lower labour requirements. So, beside its higher environmental impacts and costs, rice systems’ intensification through irrigation might not be the way chosen by the farmers.

5. Conclusion

This research has implemented a joint approach of techno-economic performances and environmental impacts in a diversity of actual cropping systems classified as wet-season rain-fed (Rw), wet-season irrigated (Iw), and dry-season irrigated systems (Id); data collected refer to 2010 cropping seasons.

According to techno-economic and environmental criteria, all results converge and establish that dry-season irrigated systems are performing less well than other systems. They use blue water (while other systems rely mostly on green water), require more energy, labour and agrochemicals, and ultimately yield lower production. As a result, gross and net incomes are lower. Although these results refer to only one year, they tend to explain why only half of irrigated land is actually cultivated during the dry season.

In addition to the conclusion related to the low performances of Id systems, we found a striking match between Rw and Iw systems. Indeed, performances of rain-fed and wet-season irrigated rice are comparable in both economic and environmental terms. The productivities of most production factors are higher in Rw systems, although Iw systems yield higher production. Yet again, it must be reiterated that 2010 was a wet year, favourable to Rw systems. Drier conditions during the wet season would most likely penalise Rw systems due to uncontrolled water supply, yielding less production.

Direct field emissions are comparable in all systems, with the noTable exception of CH_4, which is markedly lower in Rw systems due to water and organic residue management. All environmental impacts are higher in Id systems, whether they are expressed per area used or per mass product.

The type of research performed here is demanding. It is multidisciplinary by nature, requires a huge primary data basis and involves complex modelling. However, such methodological combination shows great potential for multi-criteria assessment of cropping systems and allows for detailed eco-efficiency analyses. Several sensitive aspects and key limitations shall be underlined and possibly addressed for future research undertaken with a similar approach.

First, sample size and sampling strategy require the utmost attention; while sample size must remain manageable (because LCA must be run on each and every unit of analysis), it should also represent the diversity of existing systems in a given area. To address this issue, the research was performed at the level of a small river basin, where rice cropping practices, if not performances, are quite homogeneous. However, the results cannot then purport to be generalisable.

Second, as demanding as it was, our data collection documented only two cropping seasons in one given year. Techno-economic and environmental performances are very dependent upon climatic conditions (through yields, water balance, growing cycle length, scheduling of field operations, etc.). Further research should address other climatic scenarios (e.g., a typical dry year, an average year, a wet year), or even better, a sequence of several years. This research was of a synchronic nature (several systems assessed at one time); further research may consider a diachronic approach (a given system assessed over several cycles).

Third, a thorough inventory cannot compensate for a lack of local references with regards to direct field emissions. In rice cropping, direct field emissions form the bulk of environmental impacts. Although ideal, field measurements (tier-3 data) are hardly feasible in conjunction with a research project such as the one performed here. However, the exclusive use of generic baseline emissions and factors (tier-1 data, such as the ones provided by IPCC) may lead to massive errors. This research tried to adapt IPCC standards and use some tier-2 information (regional data, compiled by Yan et al, 2003a; 2003b); it also attempted to more accurately model emissions to water.

Fourth, results on eco-efficiency are presented per impact category; eight eco-efficiency indicators are calculated and shown for each system. Such profusion is difficult to communicate for decision- and policy-making purposes, especially when ambiguous results or interpretation occur or when EE indicators on a given system show contradicting results. Trade-offs and possibly weighting and normalisation of the impacts are needed. Further research should investigate the development of a single EE index per system for synoptic information of decision-makers, local communities and the general public, following the model of Eco Indicator 99 for single-score environmental impacts (ecopoints). Choices have to be discussed and negotiated with these stakeholders in terms of the selection and weighting of impacts and normalisation.

6. References

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Table 4a. Production factor use and techno-economic performances per area cultivated in selected rice cropping systems of Lam Sieo Yai basin – year 2010

Production factors and performances	Reference Unit	Rain-fed			Wet-season irrigated rice			Dry-season irrigated rice		
		Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
		Ref. Unit/ha								
Land	Ha	1	1	1	1	1	1	1	1	1
Labour	man hr.	8.494	6.625	5.677	15.234	11.949	8.013	16.447	16.447	11.250
Fertiliser	kg of fertiliser	625.00	625.00	625.00	687.50	687.50	687.50	687.50	687.50	687.50
Pesticide	kg of active matter	5.070	5.070	5.070	7.356	7.356	7.356	11.575	11.575	11.575
Total water	m^3	7,866	7,401	7,401	7,866	7,401	7,401	8,119	7,307	7,306
Green water	m^3	7,401	7,401	7,401	7,401	7,401	7,401	1,916	1,916	1,916
Blue water	m^3	465	0.25	0.29	465	0.24	0.20	6,203	5,391	5,391
Total energy	MJ	17,360	17,281	17,222	19,590	19,530	19,388	20,846	19,783	18,327
Production cost	THB	20,868	20,843	20,822	22,435	22,354	22,243	23,415	22,943	20,884
Gross income	THB	28,521	27,095	22,817	33,512	29,947	27,808	30,000	26,250	22,500
Net income	THB	7,653	6,252	1,995	11,077	7,593	5,565	6,585	3,307	1,616

Note: THB = Thai Baht, currency of Thailand = approximately 0.033 US$ at the time of data collection (2010)

Table 4b. Production factors' productivities and techno-economic performances in selected rice cropping systems of Lam Sieo Yai basin – year 2010

Production factors and performances	Reference Unit	Rain-fed			Wet-season irrigated rice			Dry-season irrigated rice		
		Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
		kg of paddy rice/Ref. Unit								
Land	Ha	2,500	2,375	2,000	2,938	2,625	2,438	2,500	2,188	1,875
Labour	man hr.	440.37	358.49	235.47	366.60	219.69	160.00	222.22	133.00	160.00
Fertiliser	kg of fertiliser	4.000	3.800	3.200	4.273	3.818	3.545	3.636	3.182	2.727
Pesticide	kg of active matter	493.10	468.44	394.48	399.32	356.84	331.35	215.98	188.98	161.99
Total water	m^3	0.32	0.32	0.27	0.37	0.36	0.33	0.31	0.30	0.26
Green water	m^3	0.34	0.32	0.27	0.40	0.36	0.33	1.31	1.14	0.98
Blue water	m^3	9,500	6,933	5.37	12,483	10,985	6.31	0.41	0.40	0.35
Total energy	MJ	0.145	0.1379	0.115	0.15151	0.13441	0.12442	0.1264	0.10494	0.1023
Production cost	THB	0.120	0.114	0.096	0.131	0.117	0.110	0.107	0.095	0.090
Gross income	THB	0.083	0.083	0.083	0.083	0.083	0.083	0.083	0.083	0.083
Net income	THB	0.629	0.310	0.274	0.348	0.287	0.229	1.160	0.662	0.380

Table 5. Direct field emissions from the paddy field of Lam Sieo Yai Basin

Direct emission		Reference Unit	Quantity (kg/1 kg of paddy Hom Mali rice)								
			Rainfed			Wet-season irrigated rice			dry-season irrigated rice		
			Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
Emission to air	Methane (CH_4)	kg CH_4	0.08670	0.07594	0.07305	0.16549	0.15856	0.14878	0.19361	0.17637	0.16308
	N_2O	kg $N\text{-}N_2O$	0.00037	0.00031	0.00030	0.00032	0.00029	0.00026	0.00041	0.00035	0.00031
	NO	kg N-NO	0.00022	0.00018	0.00017	0.00018	0.00017	0.00015	0.00024	0.00021	0.00018
	NH_3	kg $N\text{-}NH_3$	0.02661	0.02241	0.02128	0.02191	0.02035	0.01818	0.02848	0.02441	0.02136
Emission to water	Nitrates	kg N	0.05042	0.04052	0.03788	0.04269	0.03876	0.03333	0.05734	0.04739	0.03993
	Phosphorus	kg P	0.01938	0.01553	0.01450	0.01898	0.01726	0.01490	0.02643	0.02194	0.01858
	Glyphosate	g	0.03834	0.03229	0.03067	0.05243	0.04869	0.04351	0.06816	0.05842	0.05112
	Calcium carbonate	g	0.03984	0.03355	0.03188	0.03269	0.03036	0.02713	0.07083	0.06071	0.05313
	Isoprocarb	g	0.00703	0.00592	0.00563	0.00577	0.00536	0.00479	0.01250	0.01071	0.00938
	Metaldehyde	g	-	-	-	0.03205	0.02976	0.02660	0.04167	0.03571	0.03125
Emission to soil	Glyphosate	g	0.03834	0.03229	0.03067	0.05243	0.04869	0.04351	0.06816	0.05842	0.05112
	Calcium carbonate	g	0.03984	0.03355	0.03188	0.03269	0.03036	0.02713	0.07083	0.06071	0.05313
	Isoprocarb	g	0.00703	0.00592	0.00563	0.00577	0.00536	0.00479	0.01250	0.01071	0.00938
	Metaldehyde	g	-	-	-	0.03205	0.02976	0.02660	0.04167	0.03571	0.03125

Table 6a. Environmental impact indicators in selected rice cropping systems of Lam Sieo Yai basin – year 2010, results expressed per ha cultivated.

Impact indicator		Reference unit	Rain-fed			Wet-season irrigated rice			dry-season irrigated rice		
			Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
			Ref. Unit/ha								
Output-related indicators	**GWP_{100}**	**kg CO_2-eq**	8,625	7,054	5,680	15,040	12,784	10,993	15,500	12,141	9,488
	EP	**kg PO_4-eq**	233	178	141	255	208	167	298	217	158
	AP	**kg SO_2-eq**	130	104	83	128	106	88	142	107	80
	ODP	**mg CFC-11-eq**	210	168	133	214	177	148	240	180	135
	FWAE	**kg 1,4-DB eq**	823	656	522	955	795	656	1,078	812	606
input-related indicators	**WU**	**m^3**	6,305	6,295	6,285	7,035	7,035	7,026	7,256	7,256	7,256
	LU	**Ha**	1	1	1	1	1	1	1	1	1
	EU	**MJ**	17,360	17,281	17,222	19,590	19,530	19,388	20,846	19,783	18,327

Table 6b. Environmental impact indicators in selected rice cropping systems of Lam Sieo Yai basin – year 2010, results expressed per kg rice produced

Impact indicator		Reference unit	Rain-fed			Wet-season irrigated rice			dry-season irrigated rice		
			Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
			Ref. Unit/1 kg of paddy rice								
Output-related indicators	GWP_{100}	kg CO_2-eq	3.450	2.970	2.840	5.120	4.870	4.510	6.200	5.550	5.060
	EP	kg PO_4-eq	0.093	0.075	0.070	0.087	0.079	0.069	0.119	0.099	0.084
	AP	kg SO_2-eq	0.052	0.044	0.042	0.044	0.040	0.036	0.057	0.049	0.043
	ODP	mg CFC-11-eq	0.084	0.071	0.067	0.073	0.068	0.061	0.096	0.082	0.072
	FWAE	kg 1,4-DB eq	0.329	0.276	0.261	0.325	0.303	0.269	0.431	0.371	0.323
input-related indicators	WU	m^3	3.153	2.646	2.518	2.886	2.676	2.395	3.870	3.317	2.902
	LU	Ha	0.00050	0.00042	0.00040	0.00041	0.00038	0.00034	0.00053	0.00046	0.00040
	EU	MJ	8.680	7.252	6.913	8.037	7.440	6.600	9.774	9.529	7.913

Table 8a. Eco-efficiency (total value product per environmental impact, as per category) of selected rice cropping systems of Lam Sieo Yai basin – year 2010

Impact indicator	Reference unit	Eco-Efficiency								
		Rain-fed			Wet-season irrigated rice			Dry-season irrigated rice		
		Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
		Baht/Ref. Unit								
GWP_{100}	**kg CO_2-eq**	4.225	4.040	3.478	2.661	2.464	2.344	2.37	2.16	1.935
EP	**kg PO_4-eq**	170.455	159.787	128.894	175.182	151.707	138.408	142.69	121.09	100.840
AP	**kg SO_2-eq**	289.157	275.229	231.660	332.410	297.030	275.862	281.69	246.41	211.268
ODP	**mg CFC-11-eq**	179.910	169.972	143.027	198.020	177.515	164.384	167.13	146.16	124.870
FWAE	**kg 1,4-DB eq**	45.977	43.636	36.697	44.610	39.867	37.037	37.15	32.52	27.907
WU	**m3**	4.766	4.534	3.806	5.010	4.484	4.157	4.135	3.62	3.10
LU	**ha**	30,000	28,500	24,000	35,250	31,500	29,250	30,000	26,250	22,500
EU	**MJ**	1.729	1.647	1.374	1.808	1.600	1.477	1.505	1.25	1.22

Table 8b. Net income per environmental impact (as per category) of selected rice cropping systems of Lam Sieo Yai basin – year 2010

Impact indicator	Reference unit	Net return to environmental impact								
		Rain-fed			Wet-season irrigated rice			Dry-season irrigated rice		
		Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.
		Baht/Ref. Unit								
GWP_{100}	**kg CO_2-eq**	0.887	0.886	0.351	0.737	0.594	0.506	0.425	0.272	0.170
EP	**kg PO_4-eq**	35.051	32.883	14.168	43.494	36.569	33.331	22.133	15.254	10.247
AP	**kg SO_2-eq**	60.375	59.100	24.034	86.688	71.600	63.247	46.370	31.041	20.230
ODP	**mg CFC-11-eq**	37.285	36.489	14.954	51.657	42.790	37.677	27.407	18.413	12.002
FWAE	**kg 1,4-DB eq**	9.572	9.362	3.821	11.639	9.610	8.488	6.125	4.097	2.668
WU	**m3**	1.216	0.995	0.316	1.574	1.081	0.791	0.907	0.456	0.223
LU	**Ha**	7,653	6,252	1,995	11,077	7,593	5,565	6,585	3,307	1,616
EU	**MJ**	0.441	0.361	0.114	0.568	0.386	0.281	0.330	0.157	0.087

Towards Improved Water and Nitrogen Management in Irrigation Sugar Cane Production: A Combined Analysis Using Crop Modelling and Life Cycle Analysis in Pongola, South Africa

M. van der Laan, A. Jumman, S.R. Perret

Short summary

The application of irrigation water and nitrogen (N) fertilizer in excess of crop demand reduces profitability and has multiple detrimental impacts on the environment. In this study, Life Cycle Assessment methodology was used to quantify the environmental benefits of improved management of water and fertilizer N by sugarcane farmers in a case study in Pongola, South Africa. Based on measured data and the DSSAT-Canegro model, a baseline scenario, representing farmer intuition-based irrigation scheduling management, and two additional scenarios in which water, and water and N were more rationally managed, were compared. Results show that improved water and N management can lead to a 20% reduction in non-renewable energy consumption per FU, with sustained or even increased yields. Total GHG emissions can potentially be reduced by 25% through more efficient water and N management. Limiting the rates of fertilizer N applied, made possible by decreasing N leaching through improved irrigation scheduling, resulted in the highest reductions for both impact categories. The eutrophication potential can potentially be reduced by 45%. While total water consumption was very similar between baseline and improved scheduling irrigation scenarios, more efficient use of rainfall was achieved through accurate scheduling, reducing river water extraction requirements. Such assessments can be used to encourage farmers to intensify management as well as to establish environmental stewardship incentive policies.

1. Introduction

Intensive crop production can result in a range of negative environmental impacts, including climate change as a result of greenhouse gas (GHG) emissions, water and air pollution, and consumption of non-renewable resources. There is a growing trend in the use of Life Cycle Assessment (LCA) to quantify the full range of potential environmental impacts of agricultural products. LCAs have also been effectively applied to compare the environmental impacts of alternate on-farm management practices (Brentrup et al., 2001; Brentrup et al., 2004b; Mouron et al., 2006; Blengini and Busto, 2009), and are an effective instrument for monitoring for any 'pollution swapping' (Thorburn and Wilkinson, 2012) or 'problem shifting' (Finnveden et al., 2009).

High growth rates and dry matter production of sugarcane warrants the application of relatively large quantities of irrigation water and nitrogen (N) fertilizer. Nonetheless, low water and N use efficiencies are often reported for this crop, and this can be attributed to poor farmer management practices in many cases. Water use efficiencies ranging widely from 7.4 to 16.9 tonnes cane per 100 mm of evapotranspiration (ET) and 0.5 to 1.9 tonnes sucrose per 100 mm of ET have been reported in the literature (Kingston, 1994; Inman-Bamber et al., 2000). Nitrogen use efficiencies around 50% are commonly observed worldwide (Meyer and Wood, 1994; Wood et al., 2010). Nitrogen fertiliser recommendations are usually not site-specific and crop uptake is affected by factors such as soil characteristics, cultural practices and the N fertilisers timing and application method. In Australia, Thorburn et al. (2011) observed that using the 'N replacement system', which aligns N applications with actual crop production, meant that average fertiliser applications can be reduced by 35% while still obtaining yields similar to those achieved with conventional fertilizer N management. N lost to the environment was estimated to be approximately 50% lower using the replacement system. Generally, pressure to reduce water consumption and environmental pollution is increasing in most sugarcane regions (Inman-Bamber et al., 2000; Hurst et al., 2004; Thorburn et al., 2011).

In cropping systems there is often a very strong interaction between water and N (Parashar et al., 1978; Ingram and Hilton, 1986; Widenfield, 1995). For example, reduced transpiration as a result of water stress has been observed to decrease N uptake. The high solubility of NO_3^- in most soils makes it susceptible to leaching; significant quantities of N can therefore be lost during periods of deep drainage potentially leading to N-limited crop growth later in the season. Anaerobic soil conditions associated with saturated soils arising from poor irrigation water management, poor drainage, or a combination of these can lead to increased denitrification gaseous N losses. The application of irrigation water and fertilizer N, especially in excess of crop demand, has an array of environmental impacts, including non-renewable energy consumption, greenhouse gas emissions and air and water pollution. The adoption of objective irrigation scheduling remains low in South Africa and internationally (Cluverwell et al., 1999; Stevens et al., 2005; et al., 2011), and N fertilizer is often applied in excess of crop demand to maximise crop yields based on farmer perception of target yields which are often over-estimated (Thorburn et al., 2011).

It is therefore generally accepted amongst agriculturalists that water and N use efficiencies can be increased through more intensive management. To some extent, escalating electricity and N costs is reviving farmers' interest in managing water and N more efficiently (Annandale et al., 2011).

South Africa is a water scarce country with irrigation using around 60% of total runoff water or just under 40% of exploiTable runoff (DWAF, 2004). This large proportional use of blue water has resulted in increasing pressure to transfer water to other sectors (Annandale et al., 2011). Sugarcane production occurs exclusively in the eastern parts of the country because of favourable soils and climate. Of the total area under cultivation, 25% is irrigated and predominantly located in the Crocodile River, Komati-Lomati River and Pongola River catchments. Emerging water quality problems have been observed for the rivers in these catchments (Van der Laan et al., 2012), and rising competition for water from domestic and industrial sectors, and pressure to increase international and environmental flow obligations means that irrigation has a key role to play in the sustainable use of water resources in these catchments.

LCA studies have been completed for sugarcane products in Australia (Reouf and Wegener, 2007), Mauritius (Ramjeawon, 2008), Cuba (Contreras et al., 2009), South Africa (Mashoko et al., 2010), and Brazil (Seabra et al., 2011). Seabra et al. (2000), Ramjeawon (2008) and Contreras et al. (2009) considered the environmental benefits of cane-derived products. Renouf et al. (2008) compared Australian sugarcane with United States corn and United Kingdom sugar beet as sources of bio-products, and Mashoko et al. (2010) conducted an LCA for the entire South African sugar industry (rainfed plus irrigated regions). Increasing eco-labelling requests from consumers, and obligations to report the environmental burden of sugar production and potential environmental benefits of sugarcane by-products can be informed using LCAs. LCAs can also be used to provide information to farmers about the causes of environmental impacts and investigate the management influence on environmental impacts, allowing farmers to focus on particularly relevant impacts (Mouron et al., 2006).

Presented here are results from a newly developed LCA framework to study and quantify the environmental benefits of improved irrigation and fertilizer N management in sugarcane produced under irrigation in Pongola, South Africa. Impact categories and classification/characterisation methodologies have been selected according to suitability for sugarcane production systems, and shortcomings and potential improvement to the framework are discussed.

2. Materials and methods

2.1 Goal and scope definition

The goal of this assessment is to develop a suitable LCA framework for sugarcane production and to apply it to quantify the environmental benefits of improved irrigation water and N fertilizer management. The analysis is based on data acquired during a combined lysimeter and field trial conducted in Pongola, South Africa (27°24'S, 31°35'E, 308 masl; mean annual temperature = 21°C, mean annual rainfall = 690 mm) from 1986-1989 (Thompson, 1991; Van der Laan et al., 2011). The system under investigation (see Figure 1) is a virtual farm representing the mainstream features and practices in the Pongola area (farm size, cropping system, equipment and practices).

Following calibration and validation, the DSSAT-Canegro model (Jones et al., 2003; Van der Laan et al., 2011) was used to simulate water use, crop yield and extractable sucrose content, and soil carbon and N dynamics over 16 seasons between 1986 and 2001 (two cycles of one plant crop followed by seven ratoon crops). The 1.8 m deep sandy clay soil classified as a high yield potential Hutton soil form (Soil Classification Working Group, 1991) [Ferric Luvisol (FAO)/Alfisol (USDA)] and is representative of around 23% of soils in the irrigated regions of the South African sugar industry (SASEX, 2000). Selected soil properties used for model initialization are presented in Table 1. The farm was assumed to have a total area of 200 ha under pivot irrigation, with 90% of the area being harvested annually. Actual weather data over the 16 seasons were mobilized in modelling. The focus of the study is on sugarcane production and the impact of agronomic management practices, so capital goods (farm facilities and infrastructure, irrigation equipment, etc.), personnel, the manufacturing of machinery, transport to the mill and further processing of sugarcane at the mill level are considered beyond the scope of this study. The functional unit (FU) is a metric ton of extractable sucrose produced leaving the farm gate (in the form of fresh, recently burnt, sugarcane stems). Norma lisation and weighting does not form part of this analysis, which provides only mid-point environmental impact indicators.

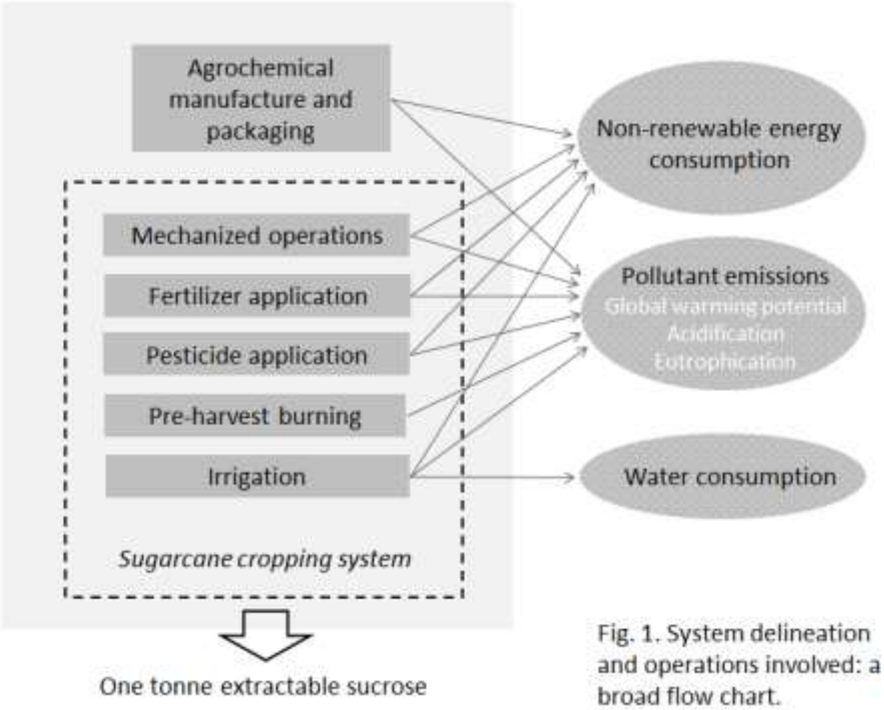

Fig. 1. System delineation and operations involved: a broad flow chart.

Table 1. Selected soil properties for the simulated Hutton soil

Depth (cm)	pH (H_2O)	Organic C (%)	Total N (%)[a]	Clay (%)	Silt (%)	Sand (%)	BD (kg m^{-3}) [b]
0-30	6.1	0.81	0.058	33	9	58	1500
30-60	5.2	0.64	0.045	43	7	50	1550
60-180	6.1	0.52	0.037	46	9	45	1500

[a] Estimated using organic C % measurement and a C:N ratio of 14:1

[b] BD = Bulk density

2.2 Management scenarios assessed

A baseline scenario was set up to represent management practices common for the region, where irrigation scheduling is often based on farmer intuition and experience rather than any scientific rationale. Irrigation applications of 12.5 mm during crop establishment and 25 mm as crop water demand increased were used. A standard scheduling programme, utilising the full application of 1000 mm per season allocated to farmers in the region, was used and no flexibility in response to rainfall events was considered.

For the second scenario (named management scenario 1), an objective irrigation scheduling scenario using a soil water depletion threshold was assessed. The management aim of this scenario was to use less irrigation water than the allocated 1000 mm by scheduling more accurately according to crop demand while maintaining or even increasing current yields. Initially, irrigation applications rates of 12.5 mm were applied during crop establishment, thereafter 25 mm applications were automatically applied when soil water was depleted by 25 mm.

For the third scenario (named management scenario 2), irrigation scheduling was determined as for management scenario 1, but seasonal N fertilizer applications were reduced to take advantage of higher soil N levels due to reduced leaching losses (resulting from improved irrigation scheduling) and to make better use of N from newly mineralized organic matter – simulated by the model to be approximately 100 kg N ha^{-1} $season^{-1}$. This is in agreement with findings by Meyer et al. (1986), who estimated soil N mineralization rates between 79 and 135 kg N ha^{-1} from the diagnostic top soil horizon of a Hutton soil form. The optimal fertilization rate of 100 kg urea-N ha^{-1} was established by iteratively using the model to identify the rate at which yields are maintained while unwanted N losses to the environment are significantly reduced.

2.3 Inventory analysis

The 200 ha farm was assumed to be irrigated by 4 x 50 ha centre pivots. Each pivot was equipped with its own dedicated pump and motor. An estimated 19.9 kW was required by the pump to meet the flow and pressure needs, and an additional 4 kW to drive the wheels, for each 50 ha centre pivot. The total power requirements for the 200 ha irrigation system was therefore 95.6 kW, or 0.48 kW ha^{-1}. Farm mechanization activities including ploughing, harrowing, ridging, verge control and agrochemical application was done using a 50 kW tractor. A 30-ton infield road haulage rig and mechanical loader was assumed to transport the cane out of the field. Fertilizer and pesticides rates were based on South African Sugarcane Research Institute recommendations for the region as provided to farmers (Table 2).

Fertilizer was applied in a single application and broadcast shortly after planting or ratooning. Pre-harvest burning was assumed to burn 80% of trash and green leaves, with the non-burnt fraction being returned to the soil. Pollutant emissions and the methodology used to estimate them are presented in Table 3.

Table 2. Seasonal agrochemical, diesel and electricity inputs for the simulated commercial farm

INPUTS	Units	Rate	Comments
Fertilizer			
Nitrogen	kg ha^{-1}	150 / 100[a]	Applied as urea
Phosphorus	kg ha^{-1}	20	Applied as super phosphate
Potassium	kg ha^{-1}	175	Applied as potassium chloride
Agricultural lime	kg ha^{-1}	0	Soil pH did not warrant lime application
Pesticides (active ingredient)			
Nematacide (plant)	kg ai[b] ha^{-1}	3	Prochloraz
Nematacide (ratoon)	kg ai ha^{-1}	7.2	Oxamyl
Herbicide (plant)	kg ai ha^{-1}	3	Diuron + Metribuzin
Herbicide (ratoon)	kg ai ha^{-1}	2.4	Diuron + Hexazinone
Insecticide	kg ai ha^{-1}	0	None used
Fungicide	kg ai ha^{-1}	0	None used
Energy/Fuel			
Diesel (machinery)	l ha^{-1}	119	Own figures (Based on SASRI, 2011)
Electricity (irrigation)	kWh ha^{-1}	2118	Own figures

[a]100 kg N ha^{-1} applied to management scenario 2, [b]ai = active ingredient

2.4 Impact assessment

The following environmental impact categories were assessed:

- Non-renewable energy consumption considered total fossil fuel energy used to produce agrochemicals and to provide on-farm energy (electricity and diesel);
- Global warming potential (GWP) was calculated according to IPCC (2006). GWP based on a 100-year time horizon is 310 for N_2O and 21 for CH_4. Indirect emissions of N_2O from atmospheric deposition of N in NO_x and NH_3 were also considered according to IPCC (2006);

Table 3. Pollutant emissions and non-renewable energy consumption for the simulated commercial farm

Emission Description	Units/ha	Baseline scenario	Manage-ment scenario 1	Manage-ment scenario 2	Emission Factor/Reference
FIELD EMISSIONS TO AIR					
N_2O (denitrification)	kg	1.63	0.90	0.49	DSSAT-Canegro (Van der Laan et al. 2011); N_2/N_2O estimated using Del Grosso et al. (2000)
N_2 (denitrification)	kg	4.26	2.33	1.27	DSSAT-Canegro (Van der Laan et al. 2011); N_2/N_2O estimated using Del Grosso et al. (2000)
NH_3 (volatilisation)	kg	0	0	0	DSSAT-Canegro (Van der Laan et al. 2011)
NO_x (nitrification + denitrification)	kg	0.75	0.75	0.5	0.005 kg/kg N applied (Ramjeawon, 2004)
Land use CO_2 emissions	kg	0	0	0	'Cropland remaining cropland' (IPCC, 2006)
Field emissions to water					
NO_3-N leaching	kg	67.9	54.7	26.4	DSSAT-Canegro (Van der Laan et al. 2011)
P leaching	g	77.8	42.5	42.5	0.18 g P/mm drainage (Thompson, 1991)
P runoff	kg	2.6	2.6	2.6	0.128 kg/kg P applied (Renouf et al. 2008)
Pesticide leaching/runoff	kg	0.14	0.14	0.14	0.015 kg/kg ai[a] (Renouf et al. 2008)
PRE-HARVEST BURNING EMISSIONS TO AIR					
CH_4	kg	47.7	50.1	50.1	0.0027 kg/kg DM[b] (Bernoux et al. 2011)
N_2O	kg	1.2	1.3	1.3	0.00007 kg/kg DM (Bernoux et al. 2011)
NH_3	kg	42.4	44.5	44.5	0.0024 kg/kg DM (EEA, 2009)
NO_x	kg	42.4	44.5	44.5	0.0024 kg/kg DM (EEA, 2009)
SO_x	kg	5.3	5.6	5.6	0.0003 kg/kg DM (EEA, 2009)
INPUT EMISSIONS TO AIR					
Urea	kg CO_2-e	3086	3086	2057	4.8 kg CO_2/kg product (Lal, 2004)
Super phosphate	kg CO_2-e	50	50	50	0.7 kg CO_2/kg product (Lal, 2004)
Potassium	kg CO_2-e	200	200	200	0.6 kg CO_2/kg product (Lal, 2004)
Lime	kg CO_2-e	0	0	0	0.12 for limestone / 0.13 for dolomite - kg CO_2/kg product (IPCC, 2006)
Nematacide	kg CO_2-e	125	125	125	18.7 kg CO_2/kg ai (Lal, 2004)
Herbicide	kg CO_2-e	57	57	57	23.1 kg CO_2/kg ai (Lal, 2004)
Insecticide	kg CO_2-e	0	0	0	18.7 kg CO_2/kg ai (Lal, 2004)
Fungicide	kg CO_2-e	0	0	0	14.3 kg CO_2/kg ai (Lal, 2004)
Diesel	kg CO_2-e	313	313	313	2.63 kg CO_2/l (Bernoux et al., 2011)
Electricity (CO_2)	kg CO_2	2076	1730	1730	0.98 kg CO_2/kWh (ESKOM, 2010)
Electricity (SO_x)	kg SO_x	17	14	14	8.1 g SO_x/kWh (ESKOM, 2010)
Electricity (NO_x)	kg NO_x	9	7	7	4.17 g NO_x/kWh (ESKOM, 2010)
Non-Renewable Energy Input					**Energy coefficient/Reference**
Urea (manufacture)	MJ	10262	10262	6841	68.41 MJ/kg urea-N (Bhat et al. 1994)
Super phosphate (manufacture)	MJ	489	489	489	6.82 MJ/kg P_2O_5 (Bhat et al. 1994)
Potassium (manufacture)	MJ	1214	1214	1214	2.88 MJ/kg K_2O (Bhat et al. 1994)
Urea (pack. and transport)	MJ	1058	1058	705	7.05 MJ/kg N (Mudahar and Hignett, 1987)
Super phosphate (pack. and transport)	MJ	597	597	597	8.33 MJ/kg P_2O5 (Mudahar and Hignett, 1987)
Potassium (pack. and transport)	MJ	2677	2677	2677	6.35 MJ/kg K_2O (Mudahar and Hignett, 1987)
Pesticide (manufacture)	MJ	1098	1098	1098	120 MJ/kg of ai (Mashoko et al. 2010)
Pesticide (pack. and transport)	MJ	27	27	27	3 MJ/kg of ai (Helsel, 1992)
Electricity	MJ	7625	6354	6354	3.6 MJ/kWh (Statistics South Africa, 2005)
Diesel	MJ	4950	4950	4950	41.6 MJ/kWh (Statistics South Africa, 2005)

[a]ai = active ingredient, [b]DM = dry matter

- Acidification (air) and eutrophication (water) potentials were calculated according to the Eco-indicator 95 approach (Figure 2) (Goedkoop, 1995);
- Water consumption based on Hoekstra et al., 2011. Blue water (surface water and groundwater that can be diverted to a range of human activities) and green water (water in the soil originating from rainfall) consumption was considered. Blue water consumption is equal to the amount irrigated (or modelled Irrigation Water Requirements IWR) and green water consumption equals cumulative evapotranspiration minus blue water consumption (or modelled total Crop Water Requirements CWR – IWR).

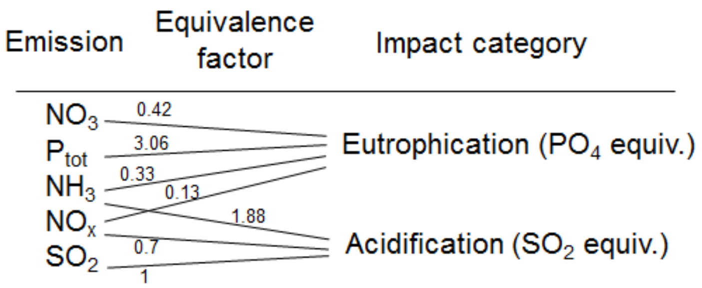

Fig. 2. Classification/characterisation of emissions using the Eco-indicator 95 method [adapted from Brentrup et al. (2001)]

3. Results and discussion

3.1 DSSAT-Canegro simulation results

Improved irrigation scheduling led to slightly increased yields and an average reduced irrigation requirement of 175 mm ha^{-1} per growing season (Table 4). Average simulated extractable sucrose mass was 17.8 t ha^{-1} for the baseline scenario and 18.5 t ha^{-1} for the scenarios in which irrigation and irrigation and N were more carefully managed. As there was no difference in yield between management scenarios 1 and 2 the estimated increase is attributed to improved irrigation water management.

Table 4. DSSAT-Canegro simulation outputs presented as growing season averages over the 16-year simulation period

Simulation Outputs	Unit	Baseline scenario	Management scenarios 1 & 2*
Green cane mass	t ha^{-1}	116.9	121.8
ExtracTable sucrose yield	t ha^{-1}	17.8	18.5
Trash mass (dry mass)	t ha^{-1}	22.1	23.2
Evapotranspiration	mm	1173	1211
Irrigation	mm	1000	825
Rainfall	mm	758	758
Deep drainage	mm	432	236
Runoff	mm	155	122

* Simulated crop yield and water balance outputs were the same for management scenarios 1 and 2

Compared to the baseline scenario, improved irrigation scheduling led to a simulated 196 mm (45%) reduction in average seasonal deep drainage and a 33 mm (21%) reduction in runoff. From Figure 3 it can, however, be observed that despite improved irrigation scheduling, high N leaching losses were still observed for management scenario 1 for certain seasons. In the final seasons of the simulation, cumulative N leaching was even higher for management scenario 1 than for the baseline scenario for two of the seasons. This is a result of the ‘carry-over’ effect, where inorganic N levels build-up in the soil as a result of previous seasons’ fertilizer applications in excess of crop demand, eventually leading to relatively high N leaching concentrations and loads. Improved irrigation scheduling as well as reducing N fertilizer application rates to 100 kg ha^{-1} led to drastic reductions in N leaching (see also

Table 3). The large seasonal variability observed for N leaching highlights the benefit of a mechanistic model to improve understanding on seasonal weather and past N fertilization management influences on N leaching, as opposed to using more simple emission factor approaches.

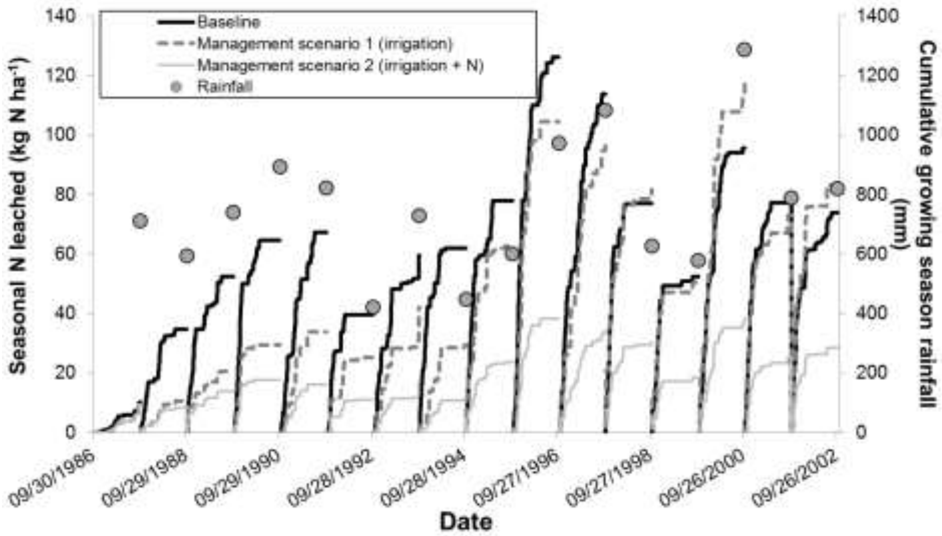

Figure 3. Simulated cumulative nitrogen (N) leached and measured cumulative rainfall over the growing season for the three management scenarios.

3.2 Interpretation of results

3.2.1 Non-renewable energy consumption

Non-renewable energy consumption ranged from 1349-1685 MJ tonne sucrose^{-1} for the three scenarios simulated (Table 5). Scheduling irrigation objectively resulted in a 132 MJ tonne sucrose^{-1} (8%) decrease in total energy input and reducing N fertilizer rates as well further reduced energy input by an additional 204 MJ tonne sucrose^{-1} (20% reduction from baseline scenario). Results are comparable with recently reported values from other sugarcane-producing regions around the world. Seabra et al. (2011) reported energy input requirements of 1109 MJ tonne sucrose^{-1} for the pre-dominantly rainfed Brazilian Centre-South Region. In comparison to our values of 257-205 MJ tonne green cane^{-1}, Renouf and Wegener (2007) reported energy inputs of 112-235 MJ tonne green cane^{-1} for rainfed sugarcane production in Queensland, Australia. Interestingly, Ricau (1980) reported much higher energy inputs for sugarcane in Louisiana in the 1970s: about 8600 MJ tonne sucrose^{-1} and 645 MJ tonne green cane^{-1}, mostly owing to much lower yields and sucrose content. He also counted the energy embedded in farm machinery (manufacturing) which constitued about 13% of all energy inputs. Such difference, however, highlights the progress made in energy use-efficiency over recent decades in cane farming and equipment application (fuel-efficient engines, lighter equipment).

Table 5. Environmental impact indicators per functional unit (FU) according to management scenario

Impact Category	Unit/FU	Baseline scenario	Management scenario 1	Management scenario 2
Energy input	MJ	1685	1553	1349
Global Warming Potential (100)	kg CO_2-e	468	421	353
Acidification potential	g SO_2-e	7.8	7.6	7.6
Eutrophication potential	g PO_4-e	16.5	9.8	9.1
Blue water consumption	m^3	562	446	446
Green water consumption	m^3	97	209	209

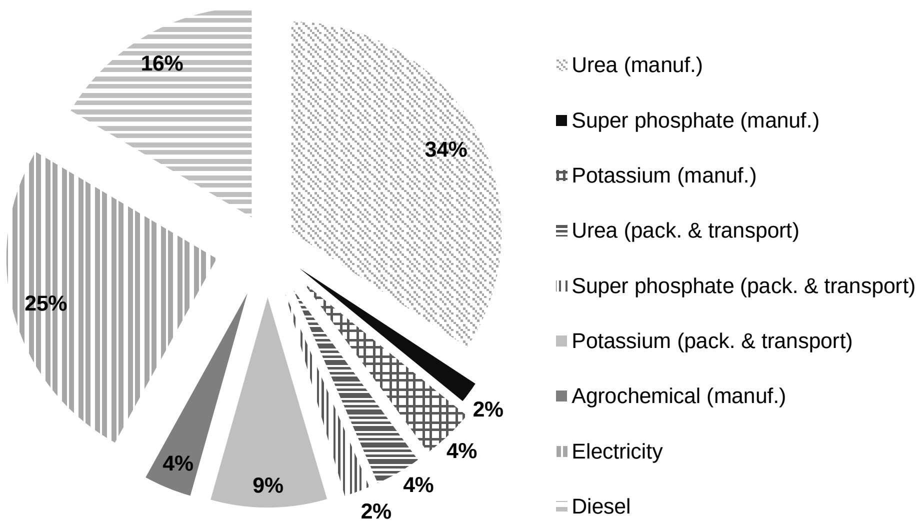

Figure 4. Relative contributions of important sugarcane production processes to non-renewable energy consumption for the baseline scenario.

The manufacture of urea was responsible for the largest fraction of total energy input requirement (34%), followed by electricity consumption for irrigation (25%) and diesel use (16%) (Figure 4). Using an ammonium-based fertilizer instead of urea at the 150 kg N ha^{-1} rate would lead to a 150 MJ tonne $sucrose^{-1}$ (9%) reduction in energy input (based on an energy coefficient of 50.6 MJ kg N^{-1}, Bhat et al. 1994), but farmers favour urea as it is significantly cheaper. Interestingly, packaging, storage and transport for the K fertilizer contributed the next highest fraction to energy input requirements for the baseline scenario at 9%, representing more than double the energy input for the manufacture of K fertilizer.

3.2.2 Global warming potential

Total greenhouse gas emissions were reduced from 468 to 421 kg CO_2-e (carbon dioxide equivalents) tonne $sucrose^{-1}$ by improving irrigation scheduling and to 353 kg CO_2-e tonne $sucrose^{-1}$ (25% decrease) by further reducing urea fertilizer application rates by 50 kg N ha^{-1}. These results compare very closely to estimates of 490 kg CO_2-e tonne sugar-1 for a study conducted in Thailand (Yuttitham et al., 2011). Emissions related to fertilizer production, storage and transport was responsible for 41% of total CO_2-e emissions per tonne of sucrose for the baseline scenario, with urea production, storage and transport alone being responsible for 37% of total CO_2-e emissions per tonne of sucrose (Figure 5). Also for the baseline scenario, electricity generation for irrigation purposes was estimated to contribute 25% of total

CO_2-e emissions per tonne of sucrose, followed closely by emissions from crop burning, which contributed 19% of total CO_2-e emissions per tonne of sucrose.

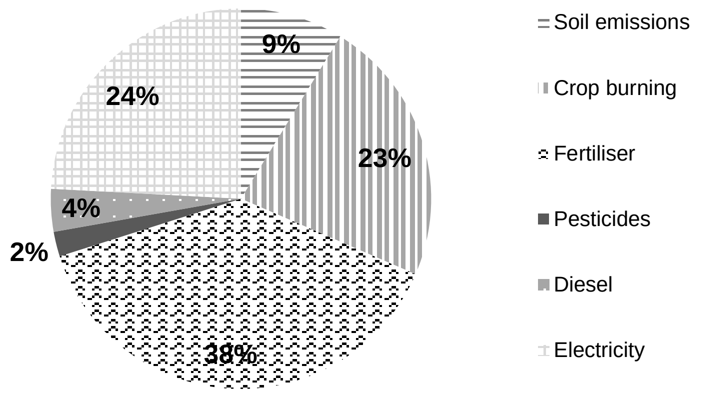

Figure 5. Relative contributions of important sugarcane production processes to total global warming potential for the baseline scenario.

Based on coefficients of variation, emission estimate uncertainty ranges of 40% for fuel use, 7% for manufacture of synthetic N fertilizers and 70% for soil N_2O emissions have been reported (Williams et al., 2006; Tuomisto et al., 2012). Using the DSSAT-Canegro model and Del Grosso et al. (2002) approach, average seasonal N_2O emissions were estimated to range from 1.63 kg ha^{-1} for the baseline scenario to 0.49 kg ha^{-1} for the scenario in which water and N were more carefully applied according to crop demand. These estimates are lower than those measured under sugarcane by Denmead et al. (2010), who observed N_2O emissions of 7.4 kg ha^{-1} on a sandy loam soil with 1.7% organic C fertilized with 150 kg urea ha^{-1}. Following a simulation study, Thorburn et al. (2010) estimated that N_2O emissions from sugarcane production for a range of environments in Australia are commonly equivalent to 3-5% of applied fertilizer N. Using the IPCC emission factor of 0.01 kg N_20-N kg fertilizer N^{-1} (IPCC, 2006) results in a N_2O emission estimate of 2.4 kg N_2O ha^{-1} for scenarios receiving 150 kg fertilizer N ha^{-1} and 1.6 kg N_2O ha^{-1} for the scenario receiving 100 kg fertilizer N ha^{-1}. We speculate that the low emission estimates in this study are a result of the system not being trash-blanketed, which potentially leads to heightened denitrification rates as a result of increased soil labile C availability to microbes and prolonged soil water saturation conditions due to reduced evaporation.

3.2.3 Eutrophication potential

The eutrophication potential was estimated at 16.5 g PO_4-e (phosphate equivalents) tonne sucrose^{-1} for the baseline scenario, 9.8 g PO_4-e tonne sucrose^{-1} for management scenario 1, and 9.1 g PO_4-e tonne sucrose^{-1} for management scenario 2. Based on seasonal averages, DSSAT-Canegro estimated that 45%, 37% and 26% of applied fertilizer N leached for the baseline, management 1 and management 2 scenarios, respectively. Using the IPCC (2006) default emission factor for N leaching of 30% of applied fertilizer N, leaching would have been under-estimated for the baseline scenario and to a lesser extent for management scenario 1, and slightly over-estimated for management scenario 2. Nitrogen loss via runoff was not considered as DSSAT-Canegro does not simulate this and there is a lack of information to estimate these losses. The P leaching factor of 0.18 g P mm drainage^{-1} was derived from lysimeter data (Thompson, 1991) and is specific to this soil and cropping system. According to Hoekstra et al. (2011), impact on water quality also needs to be considered from a geographic perspective, including the effects on seasonal water flow and quality in the specific catchment. The deterioration of water quality observed for the Pongola River is a concern (Van der Laan et al., 2012), and improving irrigation scheduling and N management can potentially reduce irrigation's contribution to pollutant loads.

3.2.4 Acidification potential

Acidification potential, resulting from the release of SO_x, NO_x and NH_3 gases during electricity generation, diesel combustion and pre-harvest burning was estimated to be 7.8 g SO_2-e tonne sucrose^{-1} for the baseline scenario and 7.6 g SO_2-e tonne sucrose^{-1} for the improved scheduling scenarios. For the baseline scenario, NH_3 and NO_x emissions from green leaf and trash burning contributed to 80% of this impact, and SO_x emissions from electricity generation for irrigation contributed 12%. Pre-harvesting burning is expected to be phased out in South Africa and the acidification potential will be subsequently reduced.

3.2.5 Water consumption

While total water consumption was very similar between the baseline and objectively irrigated scenarios (659 m^3 and 655 m^3 tonne sucrose^{-1}, respectively), blue and green water consumption differed notably between scenarios. Green water consumption for the improved scheduling scenarios (209 m^3 tonne sucrose^{-1} or 32% of total consumption) was more than double that of the baseline scenario (97 m^3 tonne sucrose^{-1} or 15% of total consumption). Increased use of green water (rainfall and soil water) is favourable as it reduces energy requirements and greenhouse gas emissions associated with irrigation, extracts less water from surface and subsurface resources, allowing for larger environmental flow, and potentially reduces pollution loads returning to the river via agricultural return flows. The water saved may then be used to irrigate larger areas of land or be diverted to other users. Of the total blue water consumption, water used during electricity generation amounted to only 0.02% of the total.

4. Improvement needs and opportunities for life cycle assessment of irrigated cropping systems

Only limited work has been done on the fate and export of pesticides from irrigated sugarcane fields at a global scale (Davis et al., 2011). Due to lack of information for South Africa on the fate of pesticides in the plant, soil, air and water compartments, or their concentration in the different parts of the plant (e.g. what is burnt, what stays in the rooting system, what is exported), it was decided to ignore human- and eco-toxicity impacts. Further work is therefore required in countries such as South Africa, Australia and India for the inclusion of these impact categories in the LCA framework.

Salt loads from irrigated lands to rivers can be substantial (Branson et al., 1975). Interestingly, salinity impacts (e.g. from irrigation return flow containing high salt loads) are not routinely considered in LCAs. This impact arises from a specific type of pollution wherein evaporation results in an increase in concentration, as opposed to the addition of a new chemical to the water (Hoekstra et al., 2011). Distinguishing between salt export resulting from irrigation activities and natural weathering is challenging. Leske and Buckley (2004) argue that salinity impacts, including impacts other than toxic effects alone, have clear cause-effect relationships between the sources and the impacts, warranting a separate salinity impact category. In water footprint assessments, Hoekstra et al. (2011) proposed the 'grey water' concept, which refers to 'the volume of freshwater that is required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality standards'. Clearly, innovative methods to quantify the impact of salinity from cropping systems within LCA frameworks are required.

Water availability for irrigation purposes is diminishing in many regions throughout the world, so region-specific approaches that take location into account and quantify the impact of water consumption on resource depletion at catchment to basin scales is needed. Pfister et al. (2009) suggested relating any water consumption calculated during LCA inventory stage to a regional Water Stress Index (WSI), so that an impact indicator – showing how impactful site-specific water consumption really is – may be derived.

Abiotic resource depletion is often considered in LCA's. Brentrup et al. (2002) question the aggregation of various resources into a single impact category, proposing instead the use of separate impact sub-categories (fossil fuels, phosphate rock and potash salt for agricultural production systems) with resources only aggregated according to primary function. Our concern with this impact category is that a production system might be 'penalised' for applying fertilizer or manure P in a particular season, while not accounting for historical P applications that have built up in the soil or the mining of soil P reserves. LCA done in any given year may, therefore, show small impacts, while actual impacts were huge in the past due to over application. A possible way to overcome this will be to rather consider P and K 'consumption', i.e. total P and K lost from the system via harvested fractions.

Land use, as an impact category, refers to the environmental impacts of occupying and utilizing land for sugar cane, with potential resultant loss of natural habitat and species diversity (Brentrup et al. 2004a). It is suggested that future studies could include all areas involved in sugar production, not only cropping areas, but also farm buildings, machinery storage areas and others, as done in mainstream LCA-based applications to farming.

5. Conclusion

Through partial application of LCA methodology, coupled with crop, nutrient, and water use modelling, the environmental impact of sugarcane produced under irrigation in South Africa for site-specific conditions has been quantified for the first time. Water and N management were shown to significantly influence a range of environmental impacts. Management practices that aim to optimise the interaction between climatic conditions and soil characteristics for a specific season to maximise crop water and N use efficiencies are essential.

In addition to irrigation water and N management, results show that discontinuing pre-harvest burning practices can significantly reduce the acidification potential impact and to a lesser extent GWP. The impacts of green cane harvesting will need to be assessed to determine whether any pollution swapping is taking place. For example, soil N_2O emissions may increase significantly as a result of the presence of a trash blanket. The importance of electricity use in GWP comes from the 'electricity mix' in South Africa, which is mostly coal-fuel burning based. Shifting trends in South African policy suggest that the environmental impact of sugarcane production could be reduced significantly not only at crop production stages but also through the co-generation of electricity using sugarcane fibre and bio-ethanol generation. Future work expanding the framework developed here to include milling processes and the benefits and impacts of mill by-products is essential. Frameworks such as these are open to further scientific progress and new information (Udo de Haes et al., 1999).

Linking with economic aspects should also be considered. This is currently challenging as in South Africa growers are no longer paid according to stalk sucrose content alone but rather according to recoverable value (RV) content, which considers sucrose, non-sucrose and fibre content. The impact of varying water and N regimes during sugarcane growth on the relative contents of all these variables in harvested cane is not yet mechanistically simulated in the DSSAT-Canegro model.

Information of this nature is anticipated to become increasingly important as a result of growing trends in the eco-labelling of food products, for example with C and water footprints. Crop modelling combined with LCA shows excellent potential in improving communication between scientists, farmers, government and non-government officials, and to assist in identifying mitigation management practices to reduce the environmental footprint of food production.

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Simplified LCA of irrigation facilities in Korea: A case study of dam and

Deuk Kim, Y., Itsubo, N. and Choi, Y.

Short summary

The aim of this study is to evaluate the environmental impacts of irrigation facilities using Life Cycle Analysis, and to compare two different types of irrigation facilities, a dam and a pumping station. LCA methodologies used in the study are CML 2 and eco-indicator 99 and used Japanese Input Output data. The functional unit is 1 cubic meter of water delivered for irrigation. This case study is a preliminary work to evaluate environmental impacts associated with agricultural water use using LCA in Korea. The study shows that the dam has less impact in the categories of global warming, acidification, eutrophication and abiotic resource depletion than the pumping station over a 70-year life time; however, results may depend upon geological condition in Korea. This case study is a challenging work to evaluate environmental impact of water supply system and to analyze inventories including water use based on.

1. Introduction

The prevailing irrigation facilities in Korea are reservoir, pumping station and weir. The number of the agricultural reservoir is 71,699 out of the 69,899 the total facilities, which is accounting for 55% of total irrigated area (MIFAFF, 2006). They have several environmental impacts in construction and operation stage. However, the environmental impact and water use in the water supply system are not properly evaluated in holistic view. The goal of the study is to evaluate the environmental impacts of two different types of water irrigation facilities using Life Cycle Assessment (LCA) tool and to compare environmental impacts and water use inventory.

Box 1. Development of National Life Cycle Inventory Database on Irrigation Water by Agricultural Dam

Young Deuk, K., Pil-Ju, P.

A study was done to develop life cycle inventory (LCI) database of dam, a major facility for irrigation water supply. The types of database developed are three out of nine dams according to the size of the water storage capacity: two kinds larger than 500,000 m3 depending on gate for discharging (Type 1) and the other dam smaller than 500,000 m3 (Type 2). According to the LCI analysis, type 1 larger than 500,000 m3 storage capacity with gate has the lowest environment impact in the 6 impact categories. The impact of the type 1 accounts for 7～35% of the type 2 for supplying irrigation water. Comparing with the environment impacts of water for other uses such as drinking and industrial water, the impacts of 1m3 irrigation water supply is 4～45% of the one for industrial water supply and 1～16% of the drinking water's. The three types of LCI DB on the irrigation water by dams will be useful in the application of Life Cycle Assessment in agricultural products and environmental labelling including carbon footprint since it complies with the guidelines of LCI DB construction issued by Ministry of Environment and Ministry of Knowledge Economy.

Full paper in Korea has been published by Journal of the Korean Society of Agricultural Engineers, (2011) 53(3): 59-64

2. Methodology

LCA of irrigation facilities has been carried out in accordance with the ISO 14044 (ISO, 2006), Inputs and outputs for raw and auxiliary materials and energy at each unit process are analyzed. Impact assessment methods are CML 2, mid-point level impact assessment method, and Eco-indicator 99(E), end-point level assessment method. Software for the assessment is Simapro 7.1, a product of Pre consultants, the Netherlands (Pre, 2009).

2.1 Scope and boundary

The reference facility is Idong dam located in Ansung city, and the comparison facility is Ensan pumping station in the same geological condition. The functional unit is defined as the size of the dam needed to supply 1,509 millions m3 of irrigation water. The system boundary was from raw materials acquisition to construction and operation and maintenance covering simplified specific cases based on the quantity data obtained from bills of quantities (BQs) of the dam and pumping station in South Korea.

Table 1. Characteristics of facilities

Class	Reference facility	Alternative facility
Name	Idong reservoir, earthfill dam	EnSan pumping station
Irrigation Area	2,156 ha	1,102 ha
Q per year	21.5MCM	11.0MCM
Capacity	Effective storage 20.9MCM	700Hp × 4EA
Construction	1964-1972	1973-1976

The system boundary of the study includes raw material acquisition and dam construction and O & M among them. External transportation and dismantling phase are not included in the whole life cycle.

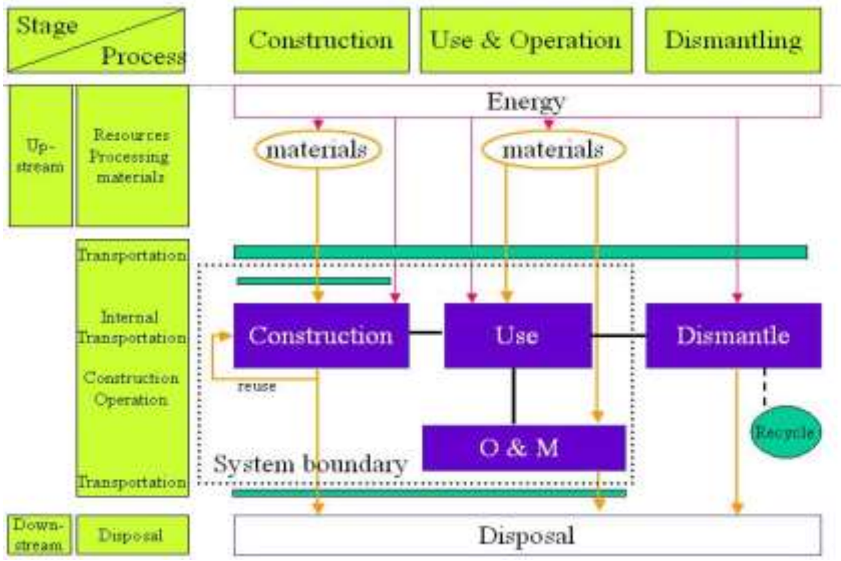

Figure 1. Simplified diagram within system boundaries for LCA of irrigation facilities

2.2 Major assumptions

99.999% cut-off rule for inclusion of inputs and outputs is applied, but raw materials that may change overall impacts were included regardless weight.

2.3 Data and quality

In the study data on processes, equipment and materials were mainly obtained from published BQs and Standard quantity estimation in civil work published by the Korean Construction Association and KRC (KCA & KRC, 2008). Most databases for construction materials, energy production and fuel consumption were based on Simapro's except electricity and cement.

Input and output data for the analysis are from the on-site for the use phase and O&M data from RIMS. Electricity for operating gate and pumps in the pumping station of use phase are considered in LCA. Data related to maintenance are obtained from the RIMS up until 2009 and estimated for the rest of life since the life span does not end.

Time-related coverage: the period of a dam construction is assumed to be 5 years and life cycle of dam 70 years.

- Geographical coverage: capital region and Gyeonggi province
- Technology coverage: Average technologies are modeled in 1970s
- Construction: calculated
- Operation: measured from 1972 to 2008 and estimated
- Maintenance: measured up until 2009 and estimated for the end-of-life data
- Electricity use: arithmetic mean of monthly data from 2007 - 2009

2.4 Procedure for calculation and water inventory

Materials (raw, auxiliary)

Conversion to mass was determined by using material density data, as data from the quantity estimation in the earthwork were given in terms of volume. Excavated soil was derived from soil tests because it is different from site to site, others from published documents. Upstream and downstream data are linked to Simapro 7.1 database.

Water use

Japanese inventory data developed by Ono (2009) has been used for calculation of water use in life cycle. The IO data is based on 2005 inter-industry relation Table the industry. For calculation of evaporation in the free water space is computed by multiplying monitoring data of evaporation in Korea Meteorological data at Suwon station by Pan Coefficient 0.7.

3. Results and discussion

3.1 Inventory Analysis

A list of inventory of dams and pumping station is shown in Table 2. It demonstrates that dominant materials in dam is concrete and steels for construction of spillway, while in pumping station concrete for building and cast iron for pump manufacturing is major input materials. Most energy for the construction is from diesel fuel rather than electricity that electricity usage is not as crucial in the inventory analysis, however; dominant energy in the operation phase of pumping station is electricity. In this study, data for the average electricity production for Korean has been used. 65% of electricity in Korea is primarily produced using coal power and nuclear power (Lee et al., 2004).

Table 2. A list of inventory of dams and pumping station Unit: Kg

In-output	**Materials**	**Dam**	**Pumping station**
Raw mat	Concrete	28,745,034	2,453,196
	Construction steel	147,718	40,130
	Cement Portland	413,330	-
	Steel bar	1,106	165
	Steel sheet	2,926	120
	Copper plate	1,137	-
	Bronze	-	4,000
	Stainless steel	-	2,000
	Cast iron GG15 I	-	202,251
Auxiliary	Diesel	620,683	967
Land	Land use (Ⅱ-Ⅲ)㎡	3,230,700	800
Energy	Electricity (Kwh/FU)	134,610	761,850
	Kwh/1000㎥	0.09	0.51
Waste	Waste oil	18,152	15
	Recycling scrap	4,861	516
	Waste wood	167,820	66,874

In terms of electricity consumption, electricity for irrigation water 1000 m^3 by dam is calculated as 0.09kwh, while water supply by pumping station needs 0.51kwh.

Figure 2 shows the water use for construction and use of irrigation facilities. As shown in the figure, 0.54L of water for 1 m^3 irrigation water from dam is needed, where 83% of water is derived from evaporation in the use phase. However, water supply from pumping station is 11 times less than dam's requirement.

3.2 Impact Assessment

Characterization result by CML 2 demonstates that AD, AP, EP, GWP, HT and PO of pumping is higher than the ones of dam. Dam has higher characterization values in the categories of TE, MAE, FWAE and ODP due to the land use and construction material use like concrete and steel. Regarding global warming potential expressed as kg CO_2 eq per ㎥, pumping station exceeding has higher score more than 49 times of dam for irrigation water supply.

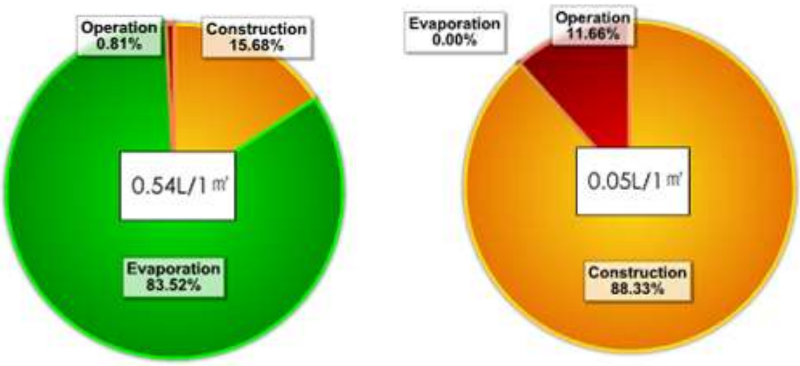

Figure 2. Water use for 1 m^3 of irrigation water in dam and pumping station in the life cycle perspectives

Figure 3 shows characterization results for 10 impact categories provided by CML 2 for the different types of irrigation facilities at the end of 70th year. According to single indicator analysis by EI 99(E), LCA of dam shows that the highest environmental impacts are associated with construction phase, accounting for more than 82.5 % of the total environmental impacts in the all categories analyzed. The stage is high for impact categories respiratory inorganic, land use, for the categories impact of the stage account for 68.3 % of total environmental impacts at construction stage. Compared to the Lee's LCA result of dam and Kim (2003) in Korea, resource uses in construction phase is a key issues in environmental burdens (Lee et al., 2008).

Regarding pumping station, the highest environmental impact is associated with operation phase, accounting for more than 94.5 % of the total environmental impacts in the all categories analyzed. The stage is high for impact categories fossil fuel, respiratory inorganic, and climate change, which reach to 93.3 % of total environmental impacts.

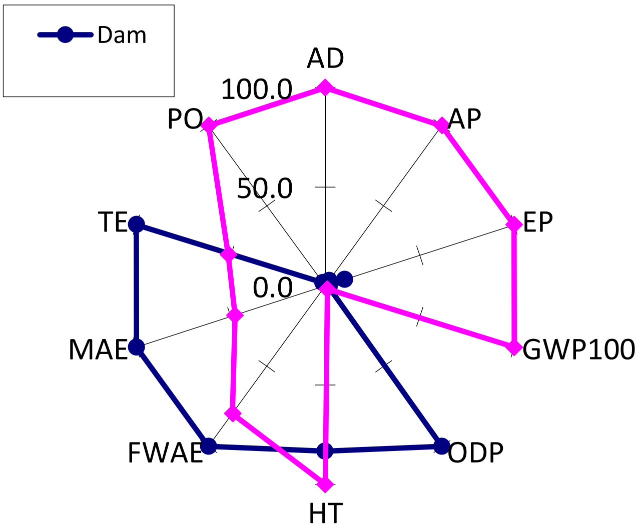

Figure 3. Characterization result of irrigation water supply

Overall, the pumping station is found to have the larger impacts per ㎥ than the dam. This is because electricity use in the pump operation for water supply has resulted in huge environmental impacts by consumption of fossil fuel and resources. Therefore, operation of the pumping station must be improved to reduce environmental impact caused by the electricity use for pumps.

It was found that most of the water consumption in life cycle of dam is generated by the evaporation. The amount of evaporation of free surface water is calculated using pan coefficient suggested by Cho (1969). Total amount of evaporation in project area for 42 year from 1967 to 2009 is 1,128 mm/yr and 0.7 was used for pan coefficient (Lee et al., 1985)[9]. He reported that evaporation from water surface is higher than precipitation.

4. Conclusions

The main findings from the comparative LCA of irrigation facilities from resource extraction to disposal can be summarized as follows:

A dominant impact comes from the construction stage in the life cycle of dam; operation stage is a main contributing phase in the life cycle of pumping station. Water supply in dam facilities used to be practiced by gravity without additional energy like electricity.

In the dam LCA it is derived from the use of construction materials in the construction stage like concrete and construction steel.

Electricity consumption is a key issue of pumping station in the operation stage due to the resource use for its generation.

Water use has been calculated using Japanese IO table, and evaporation in the water use of the dam life cycle is attributed to 83.5% of total water consumption.

There is a trade-off between dam and pumping station. Pumping station has less value in the categories of ecotoxicity and ozone depletion than dam, while the latter has less impact of global warming, acidification, eutrophication and abiotic resource depletion than the former in the 70-year life cycle of this study excluding ecological impact, depending on geological condition in Korea.

The study focus on the construction and operation stage but does not cover ecological impacts (e.g. biodiversity loss) and transportation of materials to the site in the impact assessment phase.

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General conclusions and recommendations

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