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Life Cycle Thinking for the environmental and financial assessment of rice management systems in the Senegal River Valley

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ABSTRACT

Rice is a staple food in Senegal, which however imports more than 70% of the rice consumed annually to meet its domestic demand. Despite governmental efforts to increase rice self-sufficiency, both rice supply and yields remain low. Senegalese farmers face challenges related to irrigation infrastructure and fertiliser access, besides those derived from climate change. This study applies Life Cycle Assessment (LCA) combined with financial Life Cycle Costing (LCC) to evaluate alternative scenarios for rice management in the Senegal River Valley and identify sustainability hotspots and potential improvements. Specifically, rice cultivation in Ross Béthio (Saint Louis, Senegal) is assessed based on the observed agricultural practices during the dry seasons of 2016 and 2017. Two scenarios capturing conventional (CONV) and intensive (INT) practices are compared to two reference scenarios (SAED scenarios) according to the recommendations of the official agricultural advisory service. The INT scenario generates the lowest impacts per kg of paddy rice in seven out of thirteen impact categories, including climate change, freshwater and marine eutrophication, ozone depletion and water scarcity. This is due to the higher yields (7.4 t ha^{-1}) relative to CONV (4.8 t ha^{-1}) and the two reference SAED scenarios (6.0 t ha^{-1}). The two latter scenarios show the lowest values in the remaining categories, although they also generate slightly lower profits than INT (138 € t^{-1} vs. 149 € t^{-1}) due to increased labour costs for additional fertilisation treatments. The results from both LCA and LCC underline the importance of increasing yields to decrease environmental impacts and production costs of rice when estimated per kg of product. Well-designed fertiliser application doses and timing and increased mechanisation can deliver further environmental benefits. Additional improvements (e.g. in irrigation, crop rotations, straw management) could be considered to promote the long-term sustainability and profitability of rice production in Senegal. LCA in combination with financial LCC is identified as a decision-support tool for evaluating the sustainability of alternative crop management practices. Life Cycle Thinking can still benefit from experiential learning based on information exchange between farmers, researchers and extension agents to contribute to a sustainable agriculture and ultimately to food security in Africa.

1. Introduction

Rice is a staple food in Africa, where around 40% of total rice consumption is imported, making population more vulnerable to price volatility and supply shortages (Seck et al., 2010). Sub-Saharan African countries are particularly import-dependent as rice consumption outpaces domestic supply despite progressive increases in rice harvested areas and yields (Diagne et al., 2013; Nasrin et al., 2015). West Africa is the region where the fastest growth in rice demand is projected, i.e. by 13% in 2027 (OECD-FAO, 2018; USDA, 2017). Innovation, policy

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support and capacity building are increasingly demanded to boost domestic rice supply and ensure both food availability and access across Sub-Saharan Africa (Africa Rice Center, 2011; Nhamo et al., 2014; Nwilene et al., 2008). However, a shift towards more integrated rice management is needed to contribute to a broader food security concept, increasing crop yields and agricultural sustainability (West et al., 2014). Promoting a sustainable food security challenges both governments and the research community to think beyond increasing food supply (Grote, 2014; Ingram, 2011). Thus, improving rice productivity and sustainability becomes strategic for African countries to meet their 2030 Sustainable Development Goals (SDGs), as it can simultaneously reduce poverty and hunger (SDG1 and SDG2), while promoting growth and employment (SDG8) and more sustainable production and consumption (SDG12).

Senegal is the third-largest rice importer in Africa after Nigeria and South Africa (FAOSTAT, 2021). Rice consumption represents around 31% of the calorie intake in Senegal, though domestic rice production only covers about 28% of the country's requirements, challenging food security (Diagne et al., 2013; FAO, 2010; Puma et al., 2015). The Senegalese government has been launching several strategies to stimulate domestic production, such as the *national programme for rice self-sufficiency* (PNAR). This was intended to increase total rice supply up to one million tonnes by 2012, by promoting the expansion of cultivated areas, investment in mechanisation and processing technologies, and the professionalisation of supply chain actors (MAER, 2021). The overall goal was not achieved, with around 470,000 tonnes produced in that very year (FAOSTAT, 2021). The second PNAR established new agricultural policies for the period 2014–2017, increasing rice production by more than twofold relative to 2012, i.e. at 1,011,269 tonnes by the end of the period (FAOSTAT, 2021). Average yields however decreased from 4.1 t ha^{-1} to 3.3 t ha^{-1} in that period, and have remained highly variable, though still higher than those in other Western African countries (FAOSTAT, 2021). Climate change is expected to negatively impact rice productivity in West Africa through droughts and higher temperatures (Van Oort and Zwart, 2018; Wassmann et al., 2009). Further institutional measures that promote technological innovation and more sustainable agricultural practices are needed to counteract these effects and secure rice availability in Senegal (Diagne et al., 2013; Krupnik et al., 2012).

The Senegal River Valley (SRV), located in the Sahel zone, is one of the major irrigated rice-producing areas in Senegal and the whole Africa (Van Oort and Zwart, 2018). In Senegal, rice is also produced in the Southern region of Casamance, although only the SRV diverts part of its rice production to other regional markets (USDA-GAIN, 2018). The favourable climate conditions and the adoption of varieties with shorter cropping cycles allow for two rice harvests per year, namely in the dry and rainy seasons (Van Oort et al., 2016). In the SRV, production areas are typically larger in the dry season, which brings fewer problems with pests and birds (Tanaka et al., 2015; USDA-GAIN, 2018). Average yields in SRV theoretically range between 5.0 and 6.0 t ha^{-1} in the rainy season and between 6.5 and 7.5 t ha^{-1} in the dry season (SAED, 2019; USDA-GAIN, 2021). Irrigated rice cultivation is particularly intensive in the use of water, posing technological challenges and ecological problems in regions vulnerable to climate change, in addition to financial burdens for small-scale and resource-poor farmers (Diagne et al., 2013; Krupnik et al., 2012; Venema and Schiller, 1995). Investment in more efficient irrigation systems is desirable to help mitigate the risks of drought and ensure a stable rice supply and associated revenues throughout the year (Paglietti and Machado-Mendes, 2016). Farmers face additional hurdles such as breakdown of fertiliser supply or lack of machinery; hence, inputs are often not conveniently applied regarding dates and doses (Diagne et al., 2013). Optimising agricultural practices by taking all these aspects into account is of paramount importance to improve the sustainability of rice cultivation in the SRV under resource constraints.

Rice production is a resource- and emission-intensive activity, causing environmental impacts both at regional and global scales, such as water depletion, land occupation, or global warming (Sporchia et al., 2021; Zhang et al., 2021). Rice production is responsible for approximately 11% of the global anthropogenic CH_4 emissions, the most significant source of greenhouse gas (GHG) emissions among food crops (Jiang et al., 2019). Life Cycle Thinking (LCT) has been proposed as a multi-dimensional framework to reduce impacts from production and consumption from a supply chain perspective, that is, by considering all processes from raw materials extraction to disposal (Guinée et al., 2011; Sonnemann et al., 2018). As for the environmental dimension, Life Cycle Assessment (LCA) applies LCT to quantify several impacts in terms of ecosystems damage, resource depletion, and human health. Life Cycle Costing (LCC) assesses the economic performance of production systems, often as costs and profits, and can be applied in combination with LCA to quantify impact abatement costs of alternative production technologies or decisions (Escobar et al., 2015; Fenollosa et al., 2014; Luo et al., 2009). Financial LCC considers all the activities that represent either direct costs to the decision-maker or make them a profit during the economic life of the investment, i.e. all the costs of fulfilling the functional unit (FU), including production, operation and disposal (Carlsson Reich, 2005; Lichtenvort et al., 2008). While the costs of environmental externalities could also be estimated, this requires the application of additional methodologies or principles. In general, only those costs that are likely to be covered by actors in the product system and within the decision-relevant timeframe should be monetised (Swarr et al., 2011). The application of LCA and financial LCC provides valuable information on the sustainability trade-offs of alternative scenarios. Yet, the number of studies that combine LCC with LCA to evaluate rice production systems is relatively limited (Arunrat et al., 2016; Jirapornvaree et al., 2021; Saber et al., 2020; Thanawong et al., 2014).

Many LCA studies focus on the environmental performance of rice cultivation, considering alternative crop management options (Bacenetti et al., 2016; Fusi et al., 2014; Harun et al., 2021; Xu et al., 2020). Nitrogen and phosphorus emissions from fertilisers application are a major contributor to eutrophication and acidification impacts associated with rice production; while also being a cause of climate change together with CH_4 emissions from flooded fields (Bacenetti et al., 2016; Hayashi et al., 2016; Thanawong et al., 2014). Hence, improving fertiliser management is crucial to simultaneously decrease impacts of climate change, acidification and freshwater eutrophication (Wang et al., 2010; Xu et al., 2013, 2020). Water scarcity impacts of rice have been less sistematically examined, as a broader consensus on the characterisation methods to be applied has only recently been reached (Boulay et al., 2018; Jolliet et al., 2018; Núñez et al., 2016). Similarly, ecosystems and human toxicity impacts have been frequently neglected in LCAs of agricultural products due to the lack of agreement on both the framework for modelling primary emissions from pesticides and characterisation factors for all chemicals involved (Fantke et al., 2018; Peña et al., 2019; Rosenbaum et al., 2015). This underlines the importance of adequately selecting the methods to estimate on-field emissions, especially for reactive N from fertilisers (e.g. NH_3 or N_2O). LCAs mostly rely on the Tier 1 approach of the IPCC (2006) guidelines, which provide default emission factors (EFs) that do not consider the influence of climate, soil characteristics, type of fertiliser and time of application or the irrigation system (Brodt et al., 2014; Cayuela et al., 2017). Perrin et al. (2014) recommend performing soil N balances combined with mechanistic and dynamic models to develop generic tools for calculating N-related emissions from agri-food production. Although data-intensive, mechanistic models are suitable tools for quantifying on-field emissions when fertiliser application is relevant for the LCA study (Andrade et al., 2021).

Improving the environmental and economic performances of rice production in the SRV is key to increase food self-sufficiency in Senegal, given the contribution of the region to the domestic rice supply. This study aims to evaluate observed and recommended rice cultivation practices in the municipality of Ross Béthio (Saint Louis, Senegal), located in the SRV, to identify sustainability hotspots and suggest improvements. To the best of the authors' knowledge, this is the first case study that combines LCA and financial LCC for the sustainability assessment of rice production in an African context. This study is the result of a 2-year research project (AD1511-UPV) funded by the Centre for Development Cooperation of the Polytechnic University of Valencia (Universitat Politècnica de València, Spain), carried out jointly with Caritas Spain and Caritas Senegal in 2016–2017. The project's goal was to improve rice production in Ross Béthio by increasing yields and economic profits while decreasing environmental impacts, in order to promote rural development and a sustainable food security in the region.

2. Methods

2.1. Life cycle assessment

2.1.1. Goal and scope definition

The goal of the LCA is to quantify the environmental and economic impacts of rice cultivation in Ross Béthio (Senegal), based on the agricultural practices applied by the Union of Women Rice Producers (UFPRRB, according to the French acronym) during the dry seasons of 2016 and 2017. The observed practices are compared to those reference practices recommended by the Society for the Management and Exploitation of the Senegal River Delta (SAED, according to the French acronym). SAED is a governmental organisation that promotes irrigated agriculture on the left margin of both the Senegal and Faleme rivers.

Ross Béthio is a municipality in the Dagana Department, in the North-Western region of Saint Louis. It is located 47 m above sea level, near the mouth of the Senegal River, where this provides water for rice irrigation. Until recently, the propagation of tidal variations in water level over large distances and the saline water intrusion into the river generated a vast area with weakly drained halomorphic soils (Isupova and Mikhailov, 2008). However, the hydrological regime of the Senegal River mouth area radically changed in 1986 after the *Diama dam* was built, forming an obstacle to both the upstream flow of seawater and tidal level variations. Despite the good quality of the water, soil salinity became a limiting factor for rice production in the area of study, negatively affecting rice yields.

The Saint Louis region has a tropical climate with an average low temperature above 18 °C throughout the year, which allows for two rice harvests per year. These correspond to the dry season (from February/March to June/July) and the rainy season (from July to November). The annual reference evapotranspiration is greater than 2500 mm (Ndiaye et al., 2020). Table 1 shows the climatic conditions in Ross Béthio (World Weather Online, 2021). The most important edaphic characteristics measured in the plots at the beginning of the project (2016) are included in Table 2. The high soil salinity stands out, as well as its variability depending on the location of the plot.

Table 1
Monthly mean climatic characteristics in Ross Béthio (Saint Louis, Senegal) for the period 2009–2021. Source: World Weather Online.

Month	Average high temp (°C)	Average low temp (°C)	Rainfall (mm)	Average number of days with rainfall
January	33	18	0.1	0
February	35	19	0.2	0
March	37	20	0.7	1
April	39	21	0.3	1
May	40	23	0.6	1
June	39	24	5.3	3
July	36	25	20.2	6
August	36	27	78.6	13
September	37	28	67.7	12
October	39	28	14.6	4
November	36	24	1	1
December	33	20	0.4	1

Table 2
Main characteristics of the top soil (0–10 cm) in Ross Béthio (Saint Louis, Senegal), based on the analysis of the plots (n = 5) where rice was cultivated in 2016 and 2017. CV: Coefficient of variation; EC: Electrical conductivity; Sd: Standard deviation.

	Mean	Min	Max	Sd	CV (%)
pH 1:2.5 water	5.81	5.60	6.08	0.20	3.48
EC 1:2.5 (dS/m)	7.16	1.84	15.16	5.26	73.55
Clay (%)	16.06	9.58	25.90	6.77	42.16
Silt (%)	38.07	29.19	46.29	7.35	19.32
Sand (%)	45.87	35.83	61.23	11.03	24.05
Organic C (g/kg)	5.81	3.83	7.02	1.20	20.66

The UFPRRB is responsible for 130 ha in total, handed over by the national government. Women are organised by simple agreement in groups of at least two people (either legal or physical), i.e. the so-called *economic interest groups* (GIE, according to the French acronym). GIEs do not have the obligation of initial capital and are flexible organisations from the legal point of view, which allows the most modest initiatives to be organised and access credit (Tarrière Diop, 1995). The UFPRRB consists of 28 GIEs and each producer receives 1 ha. Typically, those plots cultivated in the rainy season lay fallow in the next dry season and vice-versa. Specifically, in the dry season of 2016, Caritas funded the cultivation of 30 ha corresponding to seven GIEs; whereas, in 2017, the overall area managed by the NGO was 15 ha. It must be noted that rice production is only assessed in these two successive dry seasons, as the crop failed in the corresponding rainy seasons due to technical problems. In April 2016, the main irrigation canal was completely dry for twelve days, exactly when plants' panicles were at the initiation stage, allowing rodents to invade the plots and suck the sap. As a result, there was hardly any rice production in the rainy season of 2016. In 2017, Caritas Spain's auditors decided to withdraw the project arguing that Ross Béthio had reached an acceptable level of development. The NGO thus reallocated the funds to a one-year project in Podor, Senegal's northernmost town, on the border with Mauritania.

The primary goal of rice production in Ross Béthio is to provide food for both self-consumption and for other regional markets, contributing to food availability (see section 1). Therefore, the FU is defined under a productive scope as 1 kg of paddy rice (with 20–24% moisture). The system boundaries include all processes involved in rice production from cradle to farm gate, i.e. before drying (Fig. 1). These refer to the following processes: the production of seeds, fertilisers, and herbicides; transport of agricultural inputs to the farms; diesel production for the irrigation pump and agricultural machinery; and on-field emissions from input application. Upstream machinery production is excluded, as machinery operations are outsourced, implying intensive use and low environmental impacts per FU.

2.1.2. Description of scenarios

Two scenarios are defined describing the actual agricultural practices implemented by UFPRRB farmers, based on surveys conducted in 2016 and 2017 (Table 3). The first scenario is referred to as *conventional* (CONV) as it reflects the average practices of the GIEs through the period of study (30 ha in 2016; 9 ha in 2017). The second scenario, *intensive* (INT), corresponds to the more intensive practices applied in 2017 by a group of farmers with 6 ha in total, where both the seed and fertiliser doses were increased. Moreover, two additional *ex-ante* scenarios according to the recommendations of the official agricultural advisory service of the SRV (SAED, 2009a,b) are taken as reference, namely SAED_2td and SAED_3td, which mainly differ in terms of fertiliser doses and number of top-dressing applications. These two scenarios were not implemented in practice but represent theoretical alternatives that are also feasible for SRV farmers, taking into account the availability of fertilisers in the region and other resource constraints. Table 3 summarises the quantity and type of inputs used in each scenario, agricultural practices, and associated yields. Table 4 shows the underlying

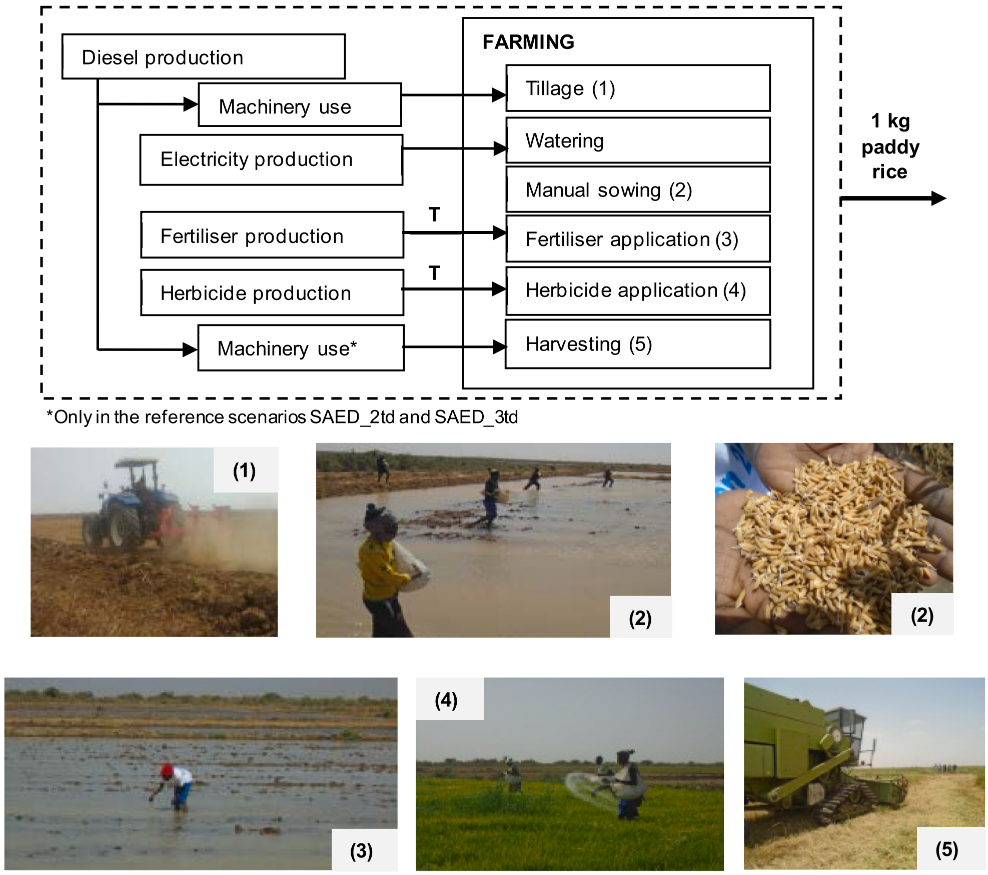

Fig. 1. System boundaries and sub-stages considered for paddy rice production in Ross Béthio (Senegal) from cradle to farm gate. Mechanical harvesting is only considered in the two recommended scenarios (SAED_2td and SAED_3td), while rice is harvested by hand in the scenarios with observed practices, namely conventional (CONV) and intensive (INT). Pictures taken and provided by technicians of Caritas Senegal. T: transport.

amount of nutrients applied per scenario.

Soil preparation is the same in all scenarios. Before sowing, deep tillage is carried out every three years with a mouldboard plough coupled to a tractor. Additionally, two weeks before sowing, a shallow tillage (10–15 cm) is made, with a disc plough followed by land levelling to improve water management and weed control. Furthermore, irrigation channels are manually cleaned and repaired at the beginning of every season.

In all scenarios, seeds of the rice variety *Sahel 108* are sown, a short-cycle variety (around 125 days) especially developed by the International Rice Research Institute to succeed under extreme conditions in the African Sahel. The major disadvantage of this variety is that it is not tolerant to salinity. Seeds are pre-germinated by soaking the seed sacks and then burying them in the ground for 24 h. Broadcast seeding is carried out by hand on irrigated plots with a 2–5 cm depth sheet of water. After sowing, fields should be periodically irrigated to hold a sheet of water of 5 cm during the vegetative phase (around 55 days). At the beginning of the reproductive phase, the water table is increased to 10 cm; then, it remains until two weeks after blossom, that is, another 55 days. Although this is the standard practice, the water table was not always maintained at the desired level, which explains the high presence of weeds observed in most plots.

As for weed control, herbicides are applied with a knapsack sprayer. The active ingredients and the corresponding doses are shown in Table 3. Following SAED's recommendations, herbicide treatment in all the scenarios consists first in spreading Bensulfuron-methyl (CAS RN 83055-99-6) on the flooded soil around six days after sowing. Propanil (CAS RN 709-98-8) and 2,4-D (CAS RN 94-75-7) are applied three weeks after sowing, after draining the soil. The plots are irrigated between 48 and 72 h later.

UFPRRB farmers do not perform a basic dressing, arguing that this later leads to more weeds. Hence, only two top dressings are applied in the CONV scenario (see Table 3): the first one with urea and diammonium phosphate (DAP) at the beginning of tillering; and the second only with urea at panicle initiation. In INT, a basic dressing is firstly applied with urea and DAP, followed by two top-dressing applications with urea. SAED_2td and SAED_3td scenarios are similar to CONV in terms of fertiliser doses, but the former use DAP for basic fertilisation and urea for top dressing. The difference between SAED_2td and SAED_3td is that urea is fractionated in two and three applications, respectively.

When rice ripening begins, plots are drained and, two weeks later, rice is harvested. In CONV and INT scenarios, rice is harvested by hand, with the help of an animal-drawn thresher. In this case, a service provider organizes a group of 6–8 workers, who harvest 2 ha per working day. Yields are estimated by each GIE based on the number of sacks obtained by each farmer and their corresponding average weight (80 kg rice per sack). Sacks that are not full are weighed *in situ*. At the end of the cropping season, the UFPRRB farmers reported average yields for the two seasons assessed of 4832 kg ha^{-1} and 7425 kg ha^{-1} for the areas under CONV and INT management, respectively. The theoretical yield in the two SAED scenarios is taken from the technical data sheet (SAED, 2009a, b). This is 6000 kg ha^{-1} assuming compliance with the input application doses and timing. SAED also recommends the use of a combine harvester for harvesting rice, which can contribute to increased productivity. Since the GIE's farmers have the possibility to outsource the use of this technology, it was considered that the two SAED scenarios employ a combine harvester with an estimated diesel consumption of 12 L ha^{-1}. While manual harvesting covers 2 ha per day, the combine harvester harvests approximately 5 ha every day, reducing labour costs.

Table 3
Agricultural inputs, machinery operations and yields considered in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td. DAP: Diammonium phosphate.

Agricultural practices	Application doses and (or) frequency	CONV	INT	SAED_2td	SAED_3td
Soil preparation	Every 3 years	Mouldboard plough coupled to tractor (89 kW/10 L diesel ha^{-1}/1 h ha^{-1})			
Basic-dressing fertilisation					
Urea	kg ha^{-1}	–	50	–	–
DAP	kg ha^{-1}	–	100	100	100
Offset	Every year	Disc plough coupled to tractor (89 kW/10 L diesel ha^{-1}/1 h ha^{-1})			
Sowing	kg ha^{-1}	120	150	120	120
Top-dressing fertilisation					
Urea	kg ha^{-1}	200 + 150	150 + 300	150 × 2	125 × 2 + 50
DAP	kg ha^{-1}	100			
Herbicides					
Bensulfuron-methyl	kg[a] ha^{-1}	0.06	0.06	0.06	0.06
2,4-D	kg[a] ha^{-1}	0.972	0.972	0.972	0.972
Propanil	kg[a] ha^{-1}	3.84	3.84	3.84	3.84
Irrigation water	m^3 ha^{-1}	17,600			
Harvesting		By hand with animal-drawn thresher (2.5 kW)		Combine harvester (95 kW/12 L diesel ha^{-1}/ 1.5 h ha^{-1})	
Paddy rice yield	kg ha^{-1}	4832	7425	6000[b]	6000[b]

[a] kg as active ingredient (a.i.).
[b] theoretical yield according to SAED (2009a, 2009b)

Table 4
Fertilisers considered and associated nutrients in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td. DAP: diammonium phosphate.

			CONV	INT	SAED_2td	SAED_3td
Basic-dressing fertilisation (kg ha^{-1})	Urea	N	–	23	–	–
	DAP	N	–	18	18	18
		P_2O_5	–	46	46	46
Top-dressing fertilisation (kg ha^{-1})	Urea	N	161	207	138	138
	DAP	N	18	–	–	–
		P_2O_5	46	–	–	–
Total nutrients (kg ha^{-1})		N	179	248	156	156
		P_2O_5	46	46	46	46

The rice straw is commonly left on the field to feed free-roaming cattle; therefore, no allocation is made between paddy rice and straw in any scenario, as the latter has no economic value.

2.1.3. Inventory data collection

In the Life Cycle Inventory (LCI) phase, several data sources were used and assumptions were made to collect data on environmentally relevant inputs and outputs from cradle to farm gate, as follows:

Agricultural inputs. Primary data on crop management and agricultural input intensity in scenarios CONV and INT were collected from surveys to UFPRRB farmers and interviews with Caritas' technicians. These data cover all agricultural practices, from soil preparation before sowing to rice harvesting. For the SAED scenarios, agricultural inputs and doses were in line with SAED's recommendations (SAED, 2009a,b), as specified above.

Emissions from fertilisers application. N losses from fertiliser application (as N_2O and NH_3 emissions to air and NO_3^- leached to groundwater) and N emission responses to soil and water management were estimated with LEACHN, the N module of the LEACHM model (Hutson and Wagenet, 1992). This a mechanistic, one-dimensional, and dynamic method in line with IPCC's Tier 3 approach, which estimates losses of ammonium, urea and nitrate by lixiviation; ammonium losses by volatilisation; and nitrate losses by denitrification. To simulate N dynamics, LEACHN models water and solute movement, as well as related chemical and biological processes in the unsaturated soil. Water and nutrient fluxes are estimated by numerical integration of the Richards' equation and the convection-dispersion equation for solute transport. Both equations can be applied to unsaturated as well as saturated soils (Hutson and Wagenet, 1992). N fluxes among compartments are simulated with first order kinetics for the N dynamics (Hutson and Wagenet, 1991). LEACHN considers three organic pools (manure, litter, and a relatively stable humus fraction) and three mineral pools (urea, ammonium, and nitrate) for N cycling. Thus, LEACHN offers advantages over IPCC's Tier 1 approach, as the former captures the influence of soil and climate conditions, together with water and N fertiliser management, on N-related emissions and the subsequent environmental impacts.

As for data requirements, LEACHN is based on actual data on N inputs through rainwater and irrigation, fertilisation and organic amendments. Nitrification, volatilisation, and denitrification rates specific for flooded rice soils were taken from Chowdary et al. (2004). Additional data on soil, irrigation and crop characteristics were obtained experimentally from one selected plot during the 2017 dry season, corresponding to the CONV scenario. In this plot, soil samples were taken before sowing at 15 cm intervals down to 30 cm (Table 5). The amount of water applied per watering in the plot was estimated by measuring the rise in the water table. Additionally, meteorological data (daily rainfall, maximum and minimum air temperature) for the seasons of study were taken from Accuweather (2017). The meteorological station in Diobène (Thiès, Senegal) was considered, as the closest one for which daily data were available. Reference evapotranspiration was estimated from the daily maximum and minimum air temperature using the Hargreaves method (Hargreaves and Samani, 1985). The hydraulic parameters of the model were estimated from the SPAW software (Saxton and Rawls, 2006) by using data on soil texture and organic carbon content from the soil analysis (see Table 5). The LEACHN model was calibrated by adjusting water infiltration and drainage according to the meteorological parameters indicated. The calibrated model was then applied to the remaining scenarios by considering the same climatic data and irrigation frequency as in CONV but varying the fertiliser application doses.

In addition to N emissions, CO_2 emissions from urea application were calculated under the Tier 1 approach of the IPCC (2006) guidelines. Emissions from phosphorus application, namely phosphate (PO_4^{3-}) to both groundwater and surface water, were calculated according to Nemecek et al. (2014). Specifically, an average P leaching of 0.07 kg ha^{-1} $year^{-1}$ and run-off of 0.175 kg ha^{-1} $year^{-1}$ was taken into account for arable land.

Methane emissions. Anaerobic decomposition of organic matter in flooded rice paddies produces CH_4, which is released to the atmosphere primarily by transport through the rice plants (IPCC, 2019). As Tier 2 EFs for CH_4 emissions specific for African countries are not available (Boateng et al., 2017), these emissions were estimated under the Tier 1 approach (IPCC, 2006), considering the revised EFs (IPCC, 2019). The following assumptions were taken: the plots are drained before the second herbicide treatment during a period shorter than three days; fields are left in fallow prior to the crop establishment, with a non-flooded pre-season longer than 180 days; and no organic matter is added.

Pesticide emissions. To estimate emissions from herbicides application, the PestLCI Consensus model v1.0 was used (Fantke et al., 2017a, b). This model departs from the original pesticide emission model PestLCI (Birkved and Hauschild, 2006; Dijkman et al., 2012). It has been further developed and parametrised to represent different crop types (including paddy rice), plant growth stages, drift deposition curves, and pesticide application methods.

Table 5
Soil properties measured in a selected plot in 2017, used as input to estimate water and nitrogen fluxes in boundary layers (soil surface and root zone bottom) in the LEACHN model.

Depth	Bulk density	Initial N–NH_4	Initial N–NO_3	Sand	Silt	Clay	Total N	Total C
(cm)	(g cm^{-3})	(kg ha^{-1})	(kg ha^{-1})	%	%	%	(g kg^{-1})	(g kg^{-1})
0–15	1.136	7.67	8.12	43.1	33.9	23.0	0.64	6.12
15–30	1.176	27.14	7.31	26.4	35.4	38.2	0.42	6.67

The primary distribution of herbicides was modelled for each scenario with the PestLCI web-tool (https://pestlciweb.man.dtu.dk/), covering the initial processes within a few minutes after application. This allowed estimating both on-field and off-field emissions in the three environmental compartments, i.e. air, water and agricultural soil. Off-field herbicide emissions (i.e. drift fraction deposited in off-field surfaces) were further distributed among the environmental compartments using the water-to-soil area ratio for Senegal. A ratio of 0.02 was estimated based on soil and land cover data at the country level (BGS, 2019; Furian et al., 2011).

The emission fractions linked to the use of herbicides are based on the respective contents of active ingredient (a.i.) and the respective application doses indicated by UFPRRB farmers and SAED recommendations. The growth stage of the plants during pesticide application was considered in combination with the BBCH-scale to determine the leaf area index, and hence, the fractions intercepted by the leaves. The BBCH-scale describes the phenological development stage for different plants, including rice. As for data requirements, the web-tool available for the PestLCI Consensus model uses actual data on pesticide treatments, considering the effect of tillage, pesticide application method, presence of buffer zones, and edaphoclimatic characteristics, among other factors.

Irrigation water. Due to lack of records, irrigation water had to be estimated in a representative plot during the dry season of 2017, based on the number of waterings made, their duration, and the pump's flow rate. The average dose of water per season was calculated at 17,600 m^3 ha^{-1}, although it can vary depending on the specific soil characteristics of the plot. The consumptive part of the irrigation water withdrawal was estimated based on the crop evapotranspiration.

Background processes. For the background LCI data, both Ecoinvent 3.6 (Wernet et al., 2016) and GaBi databases (Sphera Solutions GmbH, 2021) were used – see Table S1 in the Electronic Supplementary Material (ESM). Default processes from Ecoinvent v3.6 were chosen to include energy production, fertiliser and herbicides manufacturing, seed production and transport processes, considering cut-off system modelling. For herbicides production, Acetamide-anilide and Sulfonyl-urea compounds were respectively considered as substitutes for propanil and bensulfuron-methyl, which belong to the same chemical group but are not included in the database. Processes on machinery operations – i.e. tractor, combine harvester and irrigation pump –, were taken from the GaBi database. Emissions from both tractor and combine harvester were calculated by adjusting default processes based on primary data on fuel consumption (L ha^{-1}) and time (h ha^{-1}) needed for tillage and harvest operations, respectively.

2.1.4. Impact assessment

The ReCiPe method (Huijbregts et al., 2016) was chosen for the Life Cycle Impact Assessment (LCIA) at the midpoint level. The following impact categories were selected for being critical for the environmental assessment of agricultural systems such as rice, which are intensive in the use of fertilisers and other resources including water: climate change (CC), as CO_2-eq. for a 100-year time horizon (kg); metal depletion (MD), as Cu-eq. (kg); fossil depletion (FD), as oil-eq. (kg); terrestrial acidification (TA), as SO_2-eq. (kg); freshwater eutrophication (FwEU), as P-eq. (kg), marine eutrophication (MEU), as N-eq. (kg); ozone depletion (OD), as CFC-11-eq. (kg); and photochemical ozone formation for humans (POF-HH) and ecosystems (POF-Ec), both as NO_x-eq (kg). Toxicity impacts, namely freshwater ecotoxicity (FwEtx), human toxicity - carcinogenic (Htx-C), and human toxicity - non-carcinogenic (Htx-NC), are quantified with the USEtox 2.0 model (Fantke et al., 2017a,b) and expressed as mass-based comparative toxic quantities for ecosystems and human health, i.e. CTUe and CTUh, respectively. These, in turn, represent an estimate of the potentially affected fraction of species and human toxicity cases, respectively, integrated over time and volume, per unit of mass of chemical emitted. Finally, the AWARE 1.2C (Boulay et al., 2018) method is employed to assess the impact of blue water scarcity (WS). Under this approach, high characterisation factors for unspecified water are chosen, yielding User Deprivation Potential (UDP) in m^3 world-eq.

2.2. Financial life cycle costing

Different approaches can be taken to carry our an LCC (Ciroth et al., 2008; Lichtenvort et al., 2008). In the present study, the economic performance is assessed through a financial LCC in which the economic value is strictly determined by the internal costs generated along the life cycle. LCC always implies the perspective of an economic agent that acts as a decision-maker (Norris, 2001). In this case, farmers are the decision-makers, who do not bear the costs of environmental pollution. Hence, further externalities from environmental impacts are not taken into account. This financial LCC consists in quantifying life cycle costs, revenues and profits per FU, from cradle to farm gate, consistent with the LCA system boundaries. Total costs were thus calculated based on the management practices and inputs applied by the UFPRRB farmers (Table 3). Specifically, the costs of rice cultivation arise from soil preparation, sowing, watering, pesticide and fertiliser application, and harvesting. As shown in Table 6, production costs include labour, fertilisers, pesticides, seeds, sacks for the harvested rice, water, and machinery use; while depreciation is excluded as machinery operations are outsourced. The respective data were gathered through the above-mentioned interviews, while prices of inputs were gathered from

Table 6
Breakdown of rice production costs in Ross Béthio (Saint Louis, Senegal) in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td; considering the 2018 average exchange rate (1 € = 655.98 CFA).

	Production factors	Unit	Price
Machinery outsourcing	Soil preparation	€ ha^{-1}	3.39
	Offset	€ ha^{-1}	10.16
	Combine harvester[a]	€ ha^{-1}	76.20
Fertilisers	Urea	€ kg^{-1}	0.25
	DAP	€ kg^{-1}	0.49
Herbicides	Bensulfuron-methyl	€ kg^{-1}	15.20
	2,4 D	€ L^{-1}	6.08
	Propanyl	€ L^{-1}	4.86
Sacks	Sacks for harvested rice	€ $unit^{-1}$	0.33
Watering	Water fee plus diesel consumption for pumping	€ ha^{-1}	8.95
Seeds	Seeds	€ kg^{-1}	0.46
Labour	Labour	€ day^{-1}	2.71
	Watering	€ day^{-1}	2.22
	Labour for manual harvesting[b]	€ day^{-1}	152.00

[a] only included in the scenarios SAED_2td and SAED_3td
[b] only included in the scenarios CONV and INT.

the UFPRRB suppliers. This allowed generating a cost-sheet database reflecting both the management practices and the products and services involved in each case.

As can be seen in Table 6, the watering costs consist of a fixed annual fee plus the costs of diesel consumed by the pump and labour. It should also be noted that outsourcing the combine harvester has the same cost per ha (76.20 € ha^{-1}) as harvesting by hand (152 € day^{-1}) since, on average, the group of workers harvests 2 ha per day.

Revenues were calculated as the yield multiplied by the average selling price of paddy rice in 2016–2017 as provided by the UFPRRB, i.e. 130 CFA francs per kg. It must be noted that, on average, around 50% of the total paddy rice production is used for self-consumption, according to the surveys among UFPRRB members. Therefore, to calculate the profits, the effect of self-consumption was included as avoided (negative) costs (i.e. by not buying white rice instead), considering that 1 kg of paddy rice yields 0.63 kg of white rice (FAO, 1992). The average retail price of white rice (270 CFA kg^{-1}) was taken from the Senegalese Office of the Food Security Commissioner (Comissariat à la Sécurité Alimentaire, 2017). Finally, the profits obtained by the farmer were calculated as the difference between the revenues and total costs. Costs, revenues, and profits are expressed in € by considering the 2018 average exchange rate (1 € = 655.9890 CFA).

3. Results

3.1. On-field emissions

The calculated on-field agricultural emissions per hectare in each scenario are shown in Table 7. CH_4 emissions from anaerobic decomposition of organic matter in flooded paddy fields constitute the largest source of GHG emissions (as CO_2-eq.), followed by N_2O and CO_2 emissions from urea application. All scenarios generate the same amount of CH_4 per hectare, as this depends on the management practices (irrigation and straw management), which are the same across scenarios. Due to differences in yields, CH_4 emissions per kg of rice are estimated at 2.7 10^{-3} kg in CONV, 1.7 10^{-3} kg in INT and 1.8 10^{-3} kg in the two SAED scenarios. These respectively represent 46%, 41% and 44% of the total on-field GHG emissions per FU. On the contrary, direct CO_2 emissions make the smallest contribution to on-field GHG emissions (as CO_2-eq.) per FU, i.e. between 3% in the two SAED scenarios and 4.5% in INT. This is mainly due to the higher dose of urea applied in the latter (see Tables 3 and 4), which yields the highest CO_2 emissions per hectare among all scenarios (see Table 7). When expressed per kg of rice, CO_2 emissions are similar in CONV and INT scenarios, at around 5.0 10^{-2} kg each. The lowest values are obtained for SAED_2td and SAED_3td, at around 3.7 10^{-2} kg CO_2, respectively.

N_2O emissions make a significant contribution to total on-field GHG emissions. These are determined by the amount of N supplied per hectare, but also by the number of applications, same as any N-related emissions. N_2O emissions per hectare are 26% higher in INT than in CONV mainly due to the effect of the residual mineral N that remained in the soil after the previous crop. Intensive fertilisation often increases the residual N, which causes emissions from N volatilisation in the next cropping season. In INT, ca. 10% of the N applied to the previous crop remained in the soil, whereas in the other scenarios all mineral N was previously absorbed by the plants. Indeed, LEACHN estimates that approximately 70% of denitrification and leaching losses in INT take place during the first 15 days after sowing. SAED_2td and SAED_3td deliver 11% and 8% lower N_2O emissions than CONV; due to the lower N doses and because top dressing is split into two and three applications, respectively. When expressed per FU, all scenarios generate N_2O emissions in similar amounts, i.e. around 2.0 10^{-3} kg per kg of paddy rice. This implies 39%, 44%, 42% and 41% of total on-field GHG emissions (as kg CO_2-eq.) in INT, CONV, SAED_2td and SAED_3td scenarios, respectively.

As for the remaining N emissions, INT generates 36% higher NH_3 emissions from ammonia volatilisation than CONV, when estimated per hectare. Lower urea application doses translate into 13% lower NH_3 emissions in SAED_2td; and 24% lower in SAED_3td, where the same urea dose is split into three applications. As observed for N_2O emissions, CONV and INT scenarios generate NH_3 emissions in similar amounts per kg of paddy rice (1.2 10^{-2} and 1.1 10^{-2} kg, respectively), whereas the lowest values are obtained for SAED_2td (8.4 10^{-3} kg) and SAED_3td (7.3 10^{-3} kg). NO_3^- emissions to water per hectare are the highest in INT, followed by the two SAED scenarios and CONV. This is related to the higher leaching rates estimated by LEACHN in INT at the beginning of the cropping season, due to residual NO_3^-. In INT, basic dressing also contributes to increased NO_3^- losses, as more N is readily available in the soil. When estimated per kg of rice, CONV generates the highest NH_3 emissions (8.9 10^{-3} kg) and INT the lowest (6.6 10^{-3} kg).

PO_4^{3-} emissions to water are the same in all scenarios (0.8 kg ha^{-1}) (Table 7). This is because the model employed to estimate these emissions (Nemecek et al., 2014) does not take into account the P_2O_5 dose applied, which is still the same across scenarios in this case study (Table 4). This is, in turn, due to the fact that soluble inorganic forms of P are strongly adsorbed by mineral surfaces or precipitate, making the dose irrelevant. When estimated per FU, INT delivers the lowest value (1.1 10^{-4} kg), followed by SAED_2td and SAED_3td (1.4 10^{-4} kg) and CONV (1.7 10^{-4} kg).

All scenarios generate the same amount of pesticide emissions per hectare, as the same treatments were applied per scenario (see Table 3). From primary distribution, pesticide emissions are mainly deposited in the soil. Emissions to air are fixed values subject to the application method, which was a knapsack sprayer in all cases. Hence, airborne emissions are 5% of the applied dose in all scenarios. Pesticide emissions to water represent 20% of the off-field emissions, while the remaining 80% are emissions to soil. The CONV scenario is associated with the highest values for all pesticide emissions per FU, while the lowest are estimated in INT. This is due to yield differences, as the herbicide treatment was the same in all scenarios.

3.2. Environmental impact results

The CONV scenario generates the greatest environmental impacts per FU across all impact categories, as shown in Table 8. This is mainly due to the lower average yield obtained in the 2016 and 2017 dry seasons, relative to the other three scenarios. The yield in INT is 54% higher than in CONV, which offsets the effect of the higher input intensity (e.g. 38.5% greater N application and 25% greater seeding rate), decreasing the values across impact categories relative to the latter scenario. Theoretical yields estimated in SAED_2td and SAED_3td are 24.2% higher than those in CONV, while the N supply is 12.8% lower, which also explains the lower impact values in the former scenarios. INT causes greater impact reductions – relative to CONV – than both SAED scenarios in CC, FwEU, MEU, OD, POF-Ec, POF-HH and WS, where the effect of yield prevails (see Table 8). On the contrary, SAED_2td and SAED_3td cause greater impact reductions than INT in FD, MD, TA, due to the lower doses of fertilisers applied; and in FwEtx, Htx-C and Htx-NC due to the slightly lower toxicity associated with the production and transport of fertilisers in comparison with INT; while the dose of pesticides applied is the same. Impact reductions with respect to CONV vary from 10.6% in OD in SAED_2td to 32.1% in POF-HH in INT. SAED_3td yields greater environmental impact reductions than SAED_2td in those impact categories where on-field N emissions from fertilisers are relevant due to the positive effect of fractioning the application of top-dressing fertiliser.

The analysis of the relative contribution of each emission source to the overall life cycle impacts shows similar patterns in all scenarios, as shown in Fig. 2. It should be noted that Fig. 2 shows total human toxicity (Htx) results as the sum of Htx-C and Htx-NC. On-field emissions from fertilisers make the greatest contribution to CC, FwEU, MEU, OD, and TA. Specifically, on-field emissions account for 98% of OD due to N_2O

Table 7
On-field emissions per hectare of rice in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td.

Compartment	Emission compounds	Unit	CONV	INT	SAED_2td	SAED_3td
to air	NH_3	kg ha^{-1}	58.0	79.2	50.6	44.1
	N_2O	kg ha^{-1}	9.7	12.3	10.8	10.5
	CO_2	kg ha^{-1}	256.7	366.7	220.0	220.0
	CH_4	kg ha^{-1}	135.6	135.6	135.6	135.6
	Besulfuron-Methyl	kg[a] ha^{-1}	0.003	0.003	0.003	0.003
	2,4-D	kg[a] ha^{-1}	0.049	0.049	0.049	0.049
	Propanil	kg[a] ha^{-1}	0.192	0.192	0.192	0.192
to water	NO_3^-	kg[a] ha^{-1}	43.0	49.2	45.6	44.3
	PO_4^{3-}	kg[a] ha^{-1}	0.8	0.8	0.8	0.8
	Besulfuron-Methyl	kg[a] ha^{-1}	0.002	0.002	0.002	0.002
	2,4-D	kg[a] ha^{-1}	0.094	0.094	0.094	0.094
	Propanil	kg[a] ha^{-1}	0.231	0.231	0.231	0.231
to soil	Besulfuron-Methyl	kg[a] ha^{-1}	0.046	0.046	0.046	0.046
	2,4-D	kg[a] ha^{-1}	0.829	0.829	0.829	0.829
	Propanil	kg[a] ha^{-1}	3.424	3.424	3.424	3.424

[a] kg as active ingredient (a.i.).

Table 8
Environmental impact results per 1 kg of paddy rice in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td. CC: climate change; FD: fossil depletion; FwEtx: freshwater ecotoxicity; FwEu: freshwater eutrophication; Htx-C: human toxicity, cancer; Htx-NC: human toxicity, non-cancer; MD: metal depletion; NEU: marine eutrophication; OD: stratospheric ozone depletion; POF-Ec: photochemical ozone formation, ecosystems; POF-HH: photochemical ozone formation, human health; TA: terrestrial acidification; WS: water scarcity.

Impact category	Absolute impact values				Relative difference with respect to CONV (%)		
	CONV	INT	SAED_2td	SAED_3td	INT	SAED_2td	SAED_3td
CC (kg CO_2-eq.)	1.98	1.45	1.63	1.61	−26.77%	−17.68%	−18.69%
FD (kg oil-eq.)	0.144	0.113	0.109	0.103	−21.81%	−24.51%	−28.51%
FwEU (kg P-eq.)	$1.02\ 10^{-4}$	$7.45\ 10^{-5}$	$7.95\ 10^{-5}$	$7.95\ 10^{-5}$	−26.96%	−22.06%	−22.06%
MEU (kg N-eq.)	$7.18\ 10^{-4}$	$5.41\ 10^{-4}$	$6.08\ 10^{-4}$	$5.93\ 10^{-4}$	−24.65%	−15.32%	−17.41%
MD (kg Cu-eq.)	$1.78\ 10^{-3}$	$1.38\ 10^{-3}$	$1.35\ 10^{-3}$	$1.35\ 10^{-3}$	−22.47%	−24.16%	−24.16%
POF-Ec (kg NO_x-eq.)	$4.46\ 10^{-3}$	$3.04\ 10^{-3}$	$3.55\ 10^{-3}$	$3.55\ 10^{-3}$	−31.84%	−20.40%	−20.40%
POF-HH (kg NO_x-eq.)	$4.43\ 10^{-3}$	$3.01\ 10^{-3}$	$3.52\ 10^{-3}$	$3.52\ 10^{-3}$	−32.05%	−20.54%	−20.54%
OD (kg CFC-11-eq.)	$2.27\ 10^{-5}$	$1.86\ 10^{-5}$	$2.03\ 10^{-5}$	$1.98\ 10^{-5}$	−18.06%	−10.57%	−12.78%
TA (kg SO_2-eq.)	$1.88\ 10^{-2}$	$1.62\ 10^{-2}$	$1.34\ 10^{-2}$	$1.20\ 10^{-2}$	−13.83%	−28.72%	−36.17%
FwEtx (CTUe)	200	161	151	151	−19.50%	−24.50%	−24.50%
Htx-C (CTUh)	$8.93\ 10^{-9}$	$7.34\ 10^{-9}$	$6.67\ 10^{-9}$	$6.67\ 10^{-9}$	−17.81%	−25.31%	−25.31%
Htx-NC (CTUh)	$6.43\ 10^{-8}$	$5.17\ 10^{-8}$	$4.80\ 10^{-8}$	$4.80\ 10^{-8}$	−19.60%	−25.35%	−25.35%
WS (m^3 world-eq.)	39.17	27.71	30.69	30.69	−29.25%	−21.65%	−21.66%

emissions; and for more than 80% of MEU and TA, regardless of the scenario, due to NH_3, NO_3^- and PO_4^{3-}. The contribution of these emissions to FwEU is around 50%, with the largest share observed in the two SAED scenarios, i.e. 56%. On-field emissions are responsible for more than 75% of CC due to CH_4, N_2O and CO_2 emissions, e.g. 76% in INT and 79% in both SAED_2td and SAED_3td.

The production of fertilisers is a significant source of impacts due to the consumption of energy and mineral resources, with the subsequent emissions. This sub-stage accounts for >20% of the impacts of FD, FwEU, FwEtx, Htx-C, and MD across scenarios. This share is consistently larger in INT than in the other scenarios, as the amount of fertilisers applied per FU is greater. As expected, WS is mainly associated with irrigation, causing 81% of the total impact in SAED_2td and SAED_3td, 78% in CONV and 72% in INT. Fertiliser production accounts for between 17% and 25% of the WS in the SAED scenarios and INT, respectively. The diesel consumption for irrigation is the primary source of POF-Ec and POF-HH, causing more than 75% of these across scenarios. Watering is also an important contributor to FD, accounting for more than 30% of it in all scenarios, with the largest share observed in SAED_2td and SAED_3td (40%). Htx-NC is mainly associated with three stages, namely the transport of agricultural inputs, watering and the production of fertilisers. All of them account for between 20% and 36% of the impact, depending on the scenario. The remaining life cycle stages make a relatively small contribution to all impact categories. Given the low level of mechanisation, the use of agricultural machinery represents less than 1% of all impacts across scenarios, including the SAED scenarios, in which a combine harvester is used.

3.3. Economic results

As observed in the environmental impact results, the comparative results from the economic assessment are greatly influenced by the yield in each scenario (Fig. 3). The CONV scenario has the lowest yield and is associated with the highest costs per FU, i.e. 0.107 € kg^{-1}. On the contrary, the costs are the lowest in INT, at 0.079 € kg^{-1}, despite the higher input intensity – both for seeds and fertilisers. Total costs are estimated at 0.090 € kg^{-1} in SAED_2td and SAED_3td, respectively. The only difference between these two is the additional labour needed for the third top dressing in the latter, which plays a minor role in the costs per FU. It should be recalled that the two SAED scenarios employ a combine harvester, which contributes to the higher yields relative to CONV. This also translates into differences in the cost distribution across scenarios. While machinery accounts for around 17% of total costs in the two SAED scenarios, this share is <3% in both CONV and INT. In contrast, labour respectively accounts for around 25% and 23% in CONV and INT, while it represents around 10% of the total costs in the two SAED scenarios. Yet, the fertiliser acquisition accounts for the largest cost share across scenarios, varying between 26% in CONV and 29% in INT. The cost share of watering varies between 21% of the costs in INT and 24% in CONV. Costs of seeds account for around 10–11% of the total costs across scenarios. The acquisition of herbicides represents the smallest shares of the production costs (<5%), followed by the acquisition of

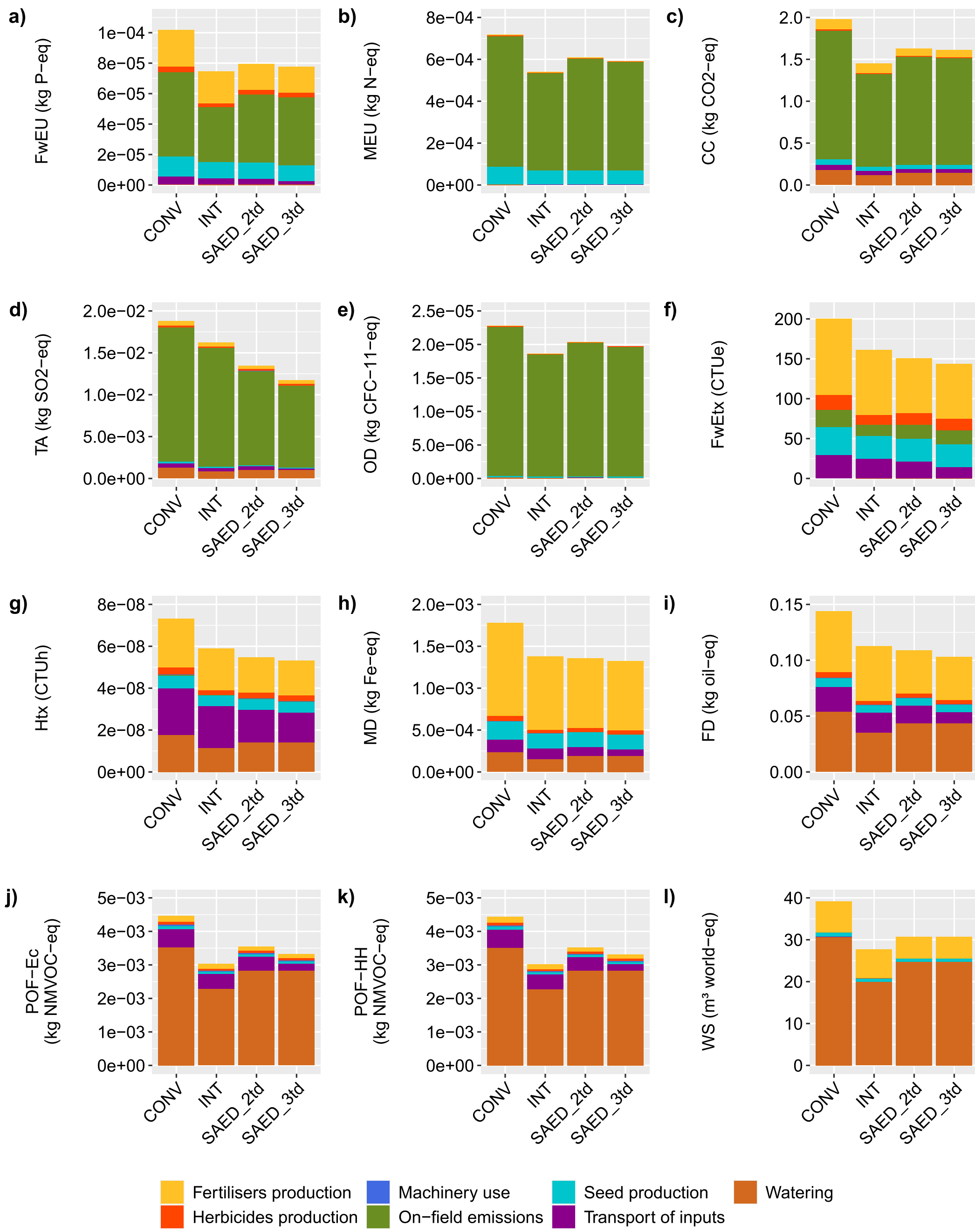

Fig. 2. Environmental impact results per 1 kg of paddy rice across impact categories, and relative contribution of the impact sources considered in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td. CC: climate change; FD: fossil depletion; FwEtx: freshwater ecotoxicity; FwEu: freshwater eutrophication; Htx: human toxicity, as the sum of carcinogenic and non-carcinogenic toxicity; MD: metal depletion; NEU: marine eutrophication; OD: stratospheric ozone depletion; POF-Ec: photochemical ozone formation, ecosystems; POF-HH: photochemical ozone formation, human health; TA: terrestrial acidification; WS: water scarcity. Figure produced in R (R Core Team, 2021), with the packages *ggplot2* (Wickham, 2016), *cowplot* (Wilke, 2020), and *scales* (Wickham, 2020).

sacks, which account for a maximum of 8% in INT.

When considering the average selling price of paddy rice and the retail price of white rice in the period of study (to include the costs avoided by self-consumption – see section 2.2), revenues per FU are estimated at 0.228 € kg^{-1} in all scenarios. It must be borne in mind that straw is not sold, which could otherwise generate different revenues per scenario due to differences in yields. Based on the above, the highest profits are obtained in the INT scenario (0.149 € kg^{-1}), followed by the two SAED scenarios (0.138 € kg^{-1}); while the lowest profits are estimated for CONV at 0.121 € kg^{-1}. Although the costs per hectare of using combine harvester are the same as those of manual harvesting (50,000 CFA ha^{-1}), mechanisation increases the cost-efficiency per unit of output, increasing profits per FU.

4. Discussion

4.1. Comparison of environmental impacts of rice cultivation in diverse locations

Several LCA case studies on the impacts of rice cultivation in multiple

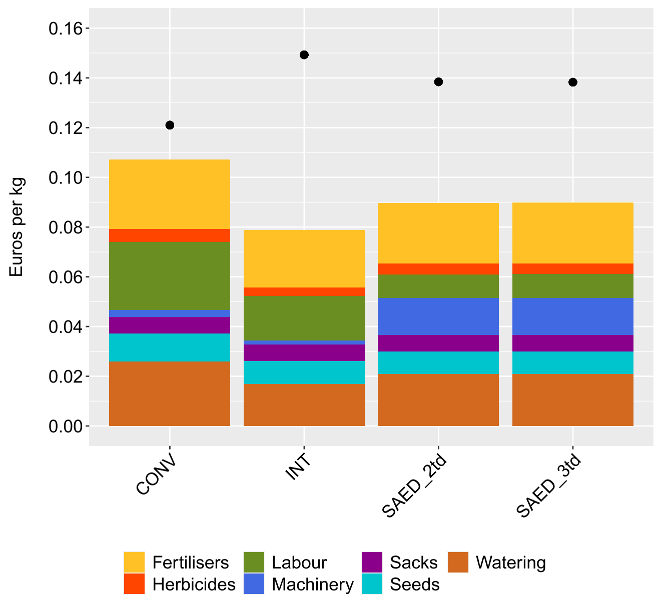

Fig. 3. Production costs, associated cost distribution and profits generated per 1 kg of paddy rice in the four scenarios evaluated: conventional (CONV), intensive (INT), and the two reference scenarios SAED_2td and SAED_3td. Black dots represent profits in each scenario, as euros per kg of paddy rice, considering the 2018 average exchange rate (1 € = 655.98 CFA). Figure produced in R (R Core Team, 2021), with the packages *ggplot2* (Wickham, 2016), *cowplot* (Wilke, 2020), and *scales* (Wickham, 2020).

regions can be found in the literature, though none of them in African countries – see Table S2 [1] in the ESM. In this section, the life cycle impacts of rice from the present study are compared to those from available literature in other geographical contexts by focusing on CC, TA and OD at the midpoint level, as the most common indicators with the same units as in this study. The impacts are discussed considering differences between the systems examined as well as other methodological choices related to the LCA application. All the LCA studies considered employ a mass-based FU, although the moisture content of the grain differs or is not always indicated. Whenever possible and for comparison purposes, yield in Table S2 is expressed as rice with 14% moisture – the recommended moisture for dried rice to avoid damage and deterioration (FAO, 1992). Impact values are thus recalculated per kg of paddy rice with the same moisture content, from cradle to farm gate only. As can be seen, many studies do not specify the rice variety, although this is a crucial factor in determining nutrient requirements and subsequent yields (Inthapanya et al., 2000).

As for the environmental evaluation, most studies focus on GHG emissions, alone (e.g. Hayashi et al., 2016; Yodkhum et al., 2017; Zhang et al., 2021) or combined with other indicators (e.g. Harun et al., 2021; Hokazono and Hayashi, 2012; Nguyen et al., 2019). All CC results are in the same order of magnitude as those obtained in this study for both CONV and INT scenarios, which are 2.1 kg CO_2-eq. and 1.6 kg CO_2-eq., respectively, per kg of rice with 14% moisture. CO_2-eq. results in the reviewed studies vary from 0.14 kg in organic production systems in Malaysia (Harun et al., 2021) to 5.6 kg in conventional irrigated systems in Thailand during the dry season (Thanawong et al., 2014). The yield considered in the former is 8 t ha^{-1} (grain moisture not specified) and around 2.2 t ha^{-1} in the latter. GHG emissions greatly depend on the fertiliser application doses and their impact on yields. Due to this interaction, changing crop management can have opposite effects. For instance, Saber et al. (2020) and Harun et al. (2021) respectively estimate twofold and threefold reductions in the carbon footprint of rice when switching from conventional to organic farming. It should be noted that Saber et al. (2020) do not specify whether they include on-field emissions of CH_4 in the LCI, nor the quantification method. In contrast, other studies show that reducing the input intensity increases GHG emissions. This is due to yield decreases, which can be >20% (Hayashi et al., 2016; Hokazono and Hayashi, 2012); or due to increases in on-field emissions from organic fertilisers (Jirapornvaree et al., 2021). The studies of conventional farms in Italy (Fusi et al., 2014) and consolidated farms in Iran (Khoshnevisan et al., 2014) present CC results below 1 kg CO_2-eq. kg^{-1}, for scenarios with relatively high yields (>5 t ha^{-1}) as combined with net N application of 82 and 203 kg ha^{-1}, respectively. The case studies in Yodkhum et al. (2017, 2018) deliver CC results around 0.6 kg CO_2-eq. kg^{-1}, with yields around 3.7 t ha^{-1} in both cases. This can be related to the share of N applied through organic fertilisers such as compost or bio-ferment juice. Khoshnevisan et al. (2014) show that increasing the level of mechanisation can have positive effects on both, yields and environmental impacts.

Incorporating the straw into the soil increases on-field CH_4 emissions and hence the contribution to CC. However, total CC values of some studies considering this practice are lower than in this study (Fusi et al., 2014; Hokazono and Hayashi, 2012; Rahman et al., 2019; Yodkhum et al., 2017). Fusi et al. (2014) consider a long fallow after straw incorporation (more than 30 days), which reduces CH_4 emissions. The authors show that straw removal requires higher mineral fertiliser doses to compensate for nutrient removal, causing increased N emissions; while CH_4 emissions are reduced because there is less organic matter in the soil. The authors also found that removing the straw could help mitigate CC and other impacts such as TA or OD. Other studies consider straw burning, assuming that it is carbon neutral (Harun et al., 2021; Hayashi et al., 2016; Jimmy et al., 2017; Saber et al., 2020). When associated emissions are included, rice straw incorporation into the soil or direct combustion for electricity generation prove better than burning in terms of CC and other environmental impacts (Yodkhum et al., 2018). Jirapornvaree et al. (2021) identify CH_4 emissions from straw residue fermentation as a GHG emission hotspot in organic rice production in Thailand. Nguyen et al. (2019) find that partial and complete removal of rice straw reduces GHG emissions by 30–40% compared to complete straw retention, based on direct field measurements in Philippines.

On-field emissions are identified as the main contributor to most environmental impacts, including CC (Fusi et al., 2014; Harun et al., 2021; He et al., 2018; Xu et al., 2020; Yodkhum et al., 2017; Zhang et al., 2021). The production of fertilisers (compost) and machinery operations can also be important impact sources (Bacenetti et al., 2016; Jimmy et al., 2017; Khoshnevisan et al., 2014). Therefore, the methods used to estimate CH_4 and N_2O emissions are particularly decisive in LCAs of rice. The IPCC Tier 1 approach is the most widely applied according to Table S2. Exceptions to this are: Hayashi et al. (2016), who measured CH_4 fluxes in experimental plots while using IPCC's Tier 2 approach for N_2O emissions; Hokazono and Hayashi (2012), who estimated N_2O emissions under IPCC's Tier 2 approach; Thanawong et al. (2014), who used Tier 2 EFs for Thailand; He et al. (2018), who used the DNDC 9.5 model for CH_4 and CO_2 emissions and N leaching (EOS, 2012); and Brodt et al. (2014), who averaged CH_4 and N_2O emission data from direct measurements in eight on-field studies in California. The latter authors highlight that their approach yields substantially lower GHG emissions than those obtained using the IPCC's Tier 1 approach. The Denitrification-Decomposition (DNDC) model (EOS, 2012) is a mechanistic model (Tier 3) that can also be used to estimate on-field emissions of paddy rice systems by incorporating several hydrological factors and irrigation information (He et al., 2018), in addition to crop-specific data

[1] The studies in Table S2 were retrieved from Scopus, by searching for the keywords life cycle assessment", "life cycle costing", "life cycle cost analysis", and "rice" in the period 2014–2021. This 8-year period covers ±4 years around the year in which the field work finished (2017). The publications were subsequently reviewed to select those that addressed irrigated paddy rice systems and could be compared to the present study, based on their goal and scope, indicators considered and other methodological aspects.

(e.g. maximum biomass or biomass C/N ratio). For instance, Lin et al. (2021) employed the DNDC model to simulate CH_4 and N_2O emissions and yields in various rotation systems with rice over different years, departing from their own measurements.

In this study, the application of LEACHN model yields higher N_2O emissions than would be obtained with the EFs from IPCC (2019). Using the Tier 1 EF for flooded paddies, 1.18 kg N_2O ha^{-1} would be released in the CONV scenario (including direct and indirect emissions), much lower than the value in Table 7. This would imply a 27% lower CC impact per FU. It must be noted that N_2O emissions are very sensitive to soil characteristics and water management, which should ideally be captured by dynamic models as the one used in this study. As a limitation, LEACHN estimates N emissions to air without differentiating between N_2O and NO_x, though the latter is not a GHG. Given the contribution of CH_4 to the impacts of paddy rice production, it is recommended to apply higher tiers for the estimation of these emissions, instead of average Tier 1 EFs, as used in the present study. Tier 2 EFs have been estimated mainly for South Eastern Asian countries, including largest rice producers (Yan et al., 2003a,b). In their meta-analysis of field measurements in the Vietnamese Mekong River Delta, Vo et al. (2020) conclude that season-based EFs could be more useful than zone-based EFs to capture variability in GHG emissions. They also show the high sensitivity of microbial CH_4 production to salinity. In the absence of Tier 2 EFs for African countries (see section 2.1.3), the use of Tier 3 mechanistic models would be desirable to capture these effects in the SRV. This involves identifying all site-specific CH_4 emission pools and influencing factors, e.g. planting dates, soil temperature and texture, water management, plant physiology and biology, etc.; with the associated data and simulation challenges (Sass and Fisher, 1997).

When looking at TA impacts, the results of this study are also of the same order of magnitude as those in Table S2. Only the study of Bacenetti et al. (2016) for organic rice stands out for the significantly higher SO_2-eq. emissions than other studies (0.2 kg per kg of paddy rice with 14% moisture), which show values in the range of 0.01–0.05 kg (Fusi et al., 2014; Jimmy et al., 2017; Thanawong et al., 2014). Bacenetti et al. (2016) consider the highest N application doses among studies (471 kg ha^{-1}), including compost. He et al. (2018) and Khoshnevisan et al. (2014) obtain lower TA values than this study, when assessing conventional and organic rice in China and Iran, respectively, despite the higher N doses applied. In the case of He et al. (2018), this can be partly explained by the relatively higher yields, i.e. between 6.2 and 8.3 t ha^{-1}. NH_3 emissions from fertilisers application are the main contributor to TA across studies, including this one. Differences in NH_3 emission estimates partly originate from the quantification method used, among other factors. OD impacts show a wide range of variability across studies, between 7.6 10^{-8} (Bacenetti et al., 2016) and 8.3 10^{-2} kg CFC-11-eq. per kg of paddy rice (Thanawong et al., 2014); again related to yield variability. The reviewed studies identify machinery use for agricultural practices as a major contributor to this impact, especially in organic production systems. OD is here quantified at 2.4 10^{-5} and 2.0 10^{-7} kg CFC-11-eq. in CONV and INT scenarios, respectively. These intermediate values are, on the one hand, related to the low mechanisation level of the practices applied by the UFPRRB and, on the other hand, to the relatively high N_2O emission intensity per hectare (see sub-sections 3.1 and 3.2). It must be noted that previous LCIA methods, like ReCiPe v1 (Huijbregts et al., 2016), did not take into account the OD potential of this substance given its different mode of action.

The results from the comparative analysis must be interpreted with care, considering differences in yields and agricultural practices, mainly N application but also other aspects such as straw management or the mechanisation level. Crop management practices influence both GHG emissions (especially CH_4) and crop yields. The challenge is to identify those practices that reduce CH_4 emissions without decreasing crop yields. In general, this can be achieved with the use of high-yielding varieties and efficient use of inputs, but actual improvements are subject to the geographic scope and local/regional conditions (Hayashi et al., 2016). The yield estimated here for the CONV scenario is very similar to that for organic rice produced in Italy (Bacenetti et al., 2016) in terms of fresh matter with 14% moisture (Table S2). While in the former 179 kg N ha^{-1} are applied as urea and DAP, the latter study considers fertilisation with green manure (133 kg N ha^{-1}) and compost (338 kg N ha^{-1}), which implies that organic N is not immediately available. Studies based in other producing regions also estimate yields around the value in CONV, despite substantial differences in management practices (e.g. Hokazono and Hayashi, 2012; Jimmy et al., 2017; Nguyen et al., 2019). Similarly, the theoretical yields in the two SAED scenarios are similar to those obtained in Malaysia (Rahman et al., 2019) or in consolidated farms in Iran (Khoshnevisan et al., 2014), with a similar N input intensity per hectare. Hayashi et al. (2016) also obtain yields around 6 t ha^{-1}, but with much lower N application doses, which corresponds to a more efficient management. In this study, the yield in the INT scenario is close to the values for conventional rice production in Italy (Fusi et al., 2014) and organic rice production in subtropical China (He et al., 2018); despite differences in fertilisation, e.g. pig manure in the latter, while in the former fertilisation is based on the nutrient removal by crop without additional nutrient application. The highest yield (9.3 t ha^{-1}) is obtained by Brodt et al. (2014) in California with only 170 kg N ha^{-1}; followed by Xu et al. (2020) and Zhang et al. (2021), both with yields >8 t ha^{-1} in Hubei Province (China).

4.2. Limitations of the study and recommendations for increasing rice sustainability in the SRV

From the results of this study, as well as from the literature review in section 4.1, several aspects can be identified as key to improve irrigated rice production under financial and logistical constraints. The timely start of the cropping season, fertiliser management, and bird damage control had been previously identified as the main factors determining rice yield in the SRV (Tanaka et al., 2015). As for fertilisation, this LCA shows that following the SAED recommendations for fertiliser application and timing could reduce environmental impacts relative to conventional practices, especially when splitting top-dressing fertilisation in three applications. This may be also beneficial from the economic point of view. Although increasing N doses usually translates into higher yields, the residual mineral N in soil also increases, potentially increasing N emissions in the next crop, as discussed in section 3.1. Among the few studies tackling the relation between N fertilization and CH_4 emissions, Linquist et al. (2012) and Zhang et al. (2014) found that the latter can increase with low mineral N application doses and vice versa. Hence, finding a balance between increased productivity and GHG emissions is recommended, ideally with moderate N application doses (Zhang et al., 2014). It would be interesting to explore other fertilisation alternatives, considering those products available in the region. For instance, ammonium sulphate has been proposed instead of urea as a source of N, as it can reduce CH_4 emissions by between 10% and 67% (Wassmann et al., 2000). Based on the latter study, this could be explained by the competition between sulphate-reducing and methanogenic bacteria. The specific GHG effects of applying ammonium sulphate should ideally be assessed with field measurements of CH_4 emissions in relation to soil and climatic conditions in the SRV; as the above-referred study, carried out in Asia.

Properly identifying weed problems is necessary to adopt effective control measures, as it was observed that the herbicides chosen in Ross-Béthio did not target all weeds. Some surveyed farmers noted that rice rotation with horticultural crops helps prevent weeds. It would also be advisable to select those herbicides with lower toxicity, which may be challenging given the limited access to the most advanced pesticide products. UFPRRB farmers and project technicians also indicated that the use of a combine harvester increases yields relative to manual harvesting because postharvest losses are reduced. This effect is captured in the two reference SAED scenarios, which are theoretically associated with higher yields than CONV. The actual yield effects of the agricultural practices recommended by the SAED, as well as their environmental and economic implications, should be better examined by implementing them *in situ*, though this was not possible in the course of this 2-year project.

Based on the literature, other options for improvement could be contemplated in the context of the SRV. Water management has been widely studied as a strategy to reduce both CH_4 emissions and irrigation water doses. Along these lines, mid-season drainage, early-season drainage, or alternative wetting and drying have shown positive effects (Islam et al., 2018). Although N_2O emissions may increase during dry periods, potential GHG benefits from reducing CH_4 normally outweigh this effect (Yan et al., 2009). Implementing intermittent irrigation in Ross Béthio may however entail technological challenges, since this requires excellent land levelling, enough water availability during critical periods of growth, good irrigation infrastructure, and efficient weed control, which may not be guaranteed in the region (Dobermann, 2004). Indeed, project technicians working with the UFPRRB farmers identified bad levelling as a problem for irrigation and mechanisation. Irrigation infrastructure is deficient across West Africa in general (Tanaka et al., 2015); and in the SRV in particular, where it requires large and efficient public irrigation schemes to be implemented (Balasubramanian et al., 2007; Krupnik et al., 2012). Soil salinity is an additional limitation for the UFPRRB to establish intermittent irrigation, as salt accumulates in dry periods, while flooding washes it. Improving irrigation infrastructure in Ross Béthio should become the priority to increase water use efficiency and reduce production costs, while delivering further environmental benefits. Controlling water regimes during fallow periods is also decisive for the environmental performance of rice cultivation. On the one hand, non-flooded preseason for more than 180 days before the cultivation period decreases CH_4 emissions (IPCC, 2019). On the other hand, non-flooded fallow can lead to salt accumulation through water evaporation from the soil surface. Balasubramanian et al. (2007) recommend mulching as a mean to reduce both evaporation and salt accumulation.

The UFPRRB uses rice straw to feed free-roaming cattle, avoiding negative environmental consequences from straw burning (Yodkhum et al., 2018). At the same time, this decreases CH_4 emissions from the organic matter decomposition that takes place when the straw is incorporated into the soil (Fusi et al., 2014), potentially increasing soil salinity. Alternative options could be explored, such as employing the straw as a renewable energy source in biogas or electricity plants, which can yield environmental and economic benefits (Prasad et al., 2020; Soam et al., 2017). However, this would require further investments as well as collaboration between supply chain actors (e.g. through social networks) to link farmers to downstream users of rice straw (Minas et al., 2020). Converting straw to biochar with domestic stoves (Mohammadi et al., 2016) or to biogas (Soam et al., 2017) have proven environmentally friendly alternatives, which could be implemented at the household scale, increasing farmers' income. Finally, biochar application to soil can improve its nutrient content, replacing fertilisers (Mohammadi et al., 2016).

Developing and tailoring varieties tolerant to soil salinity is crucial given the prevailing soil characteristics in the SRV (WARDA, 2002). In irrigated areas, it is also important to evaluate the implications of choosing between shorter duration varieties that allow for two or even three harvests per year or longer duration ones that yield more per individual crop (Van Oort et al., 2016). In the Vietnamese Mekong delta floodplains, Tran et al. (2018) found that doing three harvests per year reduces soil fertility in the long term, which translates into greater fertiliser and pesticide requirements and higher production costs. Shifting to more diversified farming systems instead of intensive monocultures is thus recommended to maintain the long-term profitability of rice production. Farmers should then choose which crops and varieties to include in their crop rotations. Introducing some local legumes increases the N content in the soil, which reduces fertiliser needs and hence costs (Arunrat et al., 2016). Crop rotations could also contribute to more diversified and healthy diets in regions as the SRV where self-consumption prevails. All these aspects should be carefully considered when defining flexible and tailor-made strategies as an ongoing process to improve the sustainability of SRV rice farms. Finally, integral management practices such as the System of Rice Intensification (SRI) could be considered. This has been proposed as an environmentally-friendly option for resource-poor farmers (Uphoff et al., 2008, 2010). The actual improvements in yields and total factor productivity brought about by the SRI in multiple geographical contexts are still under scrutiny (McDonald et al., 2006; Chapagain et al., 2011; Berkhout et al., 2015). This system is labour-intensive and requires special techniques and other resources that are not readily available for farmers in the SRV, such as transplanting of young seedlings with much lower plant densities than usual, the use of high amounts of organic amendments, mechanical weed control or alternate wetting and drying (AWD).

5. Conclusions

This study combines LCA and financial LCC to assess the sustainability of rice production in the region of Saint Louis in Senegal, which is highly import-dependent, like most Sub-Saharan African countries. Specifically, rice cultivation in the SRV during the dry seasons of 2016 and 2017 is evaluated, based on observed practices in the municipality of Ross Béthio. The LCI was gathered through a cooperation project with Caritas Spain and Caritas Senegal, aiming to promote sustainable food security in the region. Two scenarios capturing conventional and intensive practices (CONV and INT) are compared to two reference scenarios theoretically defined according to the recommendations of the official agricultural advisory (SAED_2td and SAED_3td). The LCA applies updated and consensus-based impact characterisation methods for WS and human and ecological toxicity, namely AWARE 1.2C (Boulay et al., 2018) and USEtox 2.0 (Fantke et al., 2017a,b). Moreover, N-related emissions are estimated at the Tier 3 level with the LEACHN model, a mechanistic model that simulates N dynamics based on site-specific soil and climate characteristics, and the number and doses of fertiliser application and watering. To increase the consistency and accuracy of the results, it would be advisable to estimate CH_4 emissions at the same level of detail, e.g. with Tier 3 mechanistic models or on-site emission measurements. The financial performance is assessed in terms of production costs, income and profits for the farmer.

The intensive practices of INT scenario lead to higher yields and generate the lowest impacts per kg of paddy rice in seven out of the thirteen impact categories assessed. Yet, increased N application may have negative implications on the next crop due to residual N in soil, which should be evaluated from a multi-year perspective. The two reference scenarios perform better in the remaining impact categories, including FwEU, Htx, and TA. Following SAED's recommendations on fertiliser application (in terms of doses and timing) and introducing mechanical harvesting show potential to increase rice yields as well as the environmental and economic sustainability with respect to conventional practices in the SRV. The results highlight the importance of implementing well-designed agricultural practices considering resource constraints in the region. In addition to limited access to machinery and modern fertilisers, farmers in the SRV face challenges associated with soil salinity and droughts. Thus, other strategic options should be considered, mainly investing in irrigation systems to reduce water losses and overall consumption, together with the use of high-yielding seeds tolerant to salinity. The introduction of crop rotations could be considered to diversify food production and ensure the long-term fertility of soil. Most importantly, financial strategies need to be in place to encourage farmers to adopt the best techniques adapted to the edaphoclimatic and socioeconomic characteristics of the production site.

LCA in combination with financial LCC is suitable for assessing the sustainability of alternative management practices; ideally over several years, to capture variability in impacts that are highly dependent on soil and climate conditions. Although farm income provides an indication of the socioeconomic performance of agricultural systems, additional indicators would be required to address the social dimension, e.g. working conditions of farmers and seasonal workers. Social LCA comes with its own challenges in terms of data collection and indicators, which may be particularly difficult to overcome in the case of smallholder agriculture in Africa. Experiential learning based on information exchange between farmers, researchers and extension agents could help inform further crop management decisions. However, research projects in West Africa face several implementation challenges that often lead to abandonment. LCT is here presented as a decision-support tool for the sustainable management of rice production systems, with the aim to contribute to the availability of staple crops in Africa. More integral strategies targeting all supply chain actors (farmers, regional traders, companies, governments, and consumers) through private-public partnerships would be desirable to boost sustainable food security in the region and effectively contribute to SDG1 and SDG2.

Credit author statement

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Declaration of competing interest

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

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Appendix A. Supplementary data

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