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Assessing the environmental impacts of rice in an anthropized Mediterranean wetland: Towards carbon farming

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ABSTRACT

Rice fields in southern Europe are mainly located in wetlands of high ecological value, such as L'Albufera, one of the most important rice-growing areas in Spain. Considering the unique characteristics of this environment, it is crucial to identify those farming practices that minimize the on-field emissions of this crop, one of the main causes of its impacts, and increase soil organic carbon to mitigate climate change. This study assesses the environmental impact of rice grown in L'Albufera from a cradle-to-farm gate perspective, considering current practices of removing straw from the field and a scenario in which the straw is chopped and incorporated into the soil. The functional unit on which the results are expressed is 1 kg of rice. Primary data were collected from a representative farm to develop the life cycle inventory. On-field emissions and annual changes in soil organic carbon were estimated using the Denitrification-Decomposition (DNDC) mechanistic model, and the results were compared with those obtained using conventional Tier 1 and Tier 2 methods. All the impact categories were evaluated following the EF v3.0 characterization method. On-field emissions estimated with DNDC show differences from those calculated by conventional methods, mainly for CH_4, NO_3^- and NH_3. The results highlight the great importance of on-field emissions on most impacts, namely climate change, freshwater and marine eutrophication, ozone depletion, terrestrial acidification, ecotoxicity, and human toxicity. The DNDC model estimates CH_4 and CO_2 emissions to increase when straw is incorporated into the soil. However, the annual change in soil organic carbon decreases by >50 %, and NH_3 and N_2O are reduced by 13 % and 11 %, respectively. Consequently, differences in impact scores are observed for acidification, terrestrial eutrophication, and particulate matter (about 12 % higher), but also for climate change (15 % lower) and photochemical ozone formation (4 % lower). The need to harmonize methodologies to estimate on-field emissions and the changes in soil organic carbon is highlighted to improve the application of life cycle assessment to agricultural systems, particularly rice cultivation. To effectively promote carbon farming practices related to straw management in paddy rice, it is necessary to conduct long-term field studies to measure on-field emissions associated with each alternative experimentally.

1. Introduction

Rice is one of the most important crops worldwide, being a staple food for more than half of the world's population. China and India are the leading producers, with 256 million tonnes of rice, representing 85 % of the world's production (World Economic Forum, 2022). In Europe, around 3 million tonnes were produced in 2021, and Italy and Spain jointly produced 50 % of the rice grown in Europe (FAOstat, 2023). According to the same source, in 2021, 8.4680 ha were devoted to rice production in Spain, producing 617,180 t.

Agriculture is responsible for a relevant part of the negative environmental impacts at the global level, causing problems such as those associated with the use of agricultural inputs, namely pesticides and fertilizers, which release emissions such as nitrous oxide, a potent

Abbreviations: SOC, Soil Organic Carbon; DNDC, Denitrification-Decomposition Model; LCA, Life Cycle Assessment; FU, Functional unit; LCI, Life Cycle Inventory; ETc, Evapotranspiration of the crop; ET_0, Reference evapotranspiration (FAO Penman-Monteith); Pe, Effective precipitation; Kc, Crop coefficient; Pt, Total monthly precipitation; RMSE, Root Mean Square Error; EF, Environmental Footprint.

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greenhouse gas or ammonia, that contributes to acidification (Bacenetti et al., 2016; Escobar et al., 2022; Thanawong et al., 2014). In addition, anaerobic decomposition of soil organic matter in flooded rice fields emits methane (CH_4), another important greenhouse gas. Globally, rice crops release about 9–11 % of greenhouse gas emissions (Smith et al., 2014). China is responsible for 35.6 % of the worldwide agricultural methane emissions, representing 5–19 % of global emissions of this greenhouse gas (Miranda et al., 2015). Farming practices that minimize on-field emissions and increase soil organic carbon (SOC) are crucial to mitigate global warming (Yin et al., 2020). In particular, methane emissions are influenced by factors specific to the copping system, such as water regime (Maris et al., 2016) or fertilizer type (Linquist et al., 2012; Zhang et al., 2014), as well as other characteristics, such as soil type, the temperature or the rice variety (Martínez-Eixarch et al., 2018). Straw management after rice harvest also has environmental implications. Burning straw is the simplest and cheapest way to manage this by-product, releasing CO_2 and other polluting gases into the environment (Brodt et al., 2014). Soil incorporation is another common practice that increases SOC and CH_4 emissions (Wu et al., 2019). Another technique favoring the accumulation of SOC in paddy fields is eliminating or reducing tillage (conservation tillage). In the latter case, the management of tillage time can be crucial to mitigate CH_4 emissions, thus favoring an increase in SOC and reducing emissions (Liu et al., 2021; Kan et al., 2023).

In southern Europe, rice is mainly grown in wetlands, including marshes and deltas, which are considered natural ecosystems with high biological productivity and conservation value due to their biodiversity. In Spain, rice fields are typically located in humid areas of high ecological value, within or near natural areas protected by national and international regulations, such as the rice fields of Valencia, Tarragona, and Seville, the main producing areas of the country. In particular, the Valencia region is one of the leading rice producers in Spain, with 110 t in 2021, representing 20 % of the rice produced in Spain (Generalitat Valenciana, 2021). Rice cultivation in this region is concentrated in L'Albufera, a coastal lagoon on the Mediterranean coast surrounded by marshes that form a unique ecosystem. It is internationally recognized for its birdlife and was declared a Natural Park in 1986 (Soria, 2006). Jégou and Sanchis Ibor (2019) highlight its unique hydrological system, with its complex mechanisms and specific agroecosystems, as the most exceptional value of L'Albufera. However, since the 1970s, this ecosystem has suffered severe degradation mainly due to sewage and agrochemical discharges, which have transformed the water from oligotrophic to hypertrophic. The toxic effect of the pesticides used in rice cultivation also threatens this ecosystem, which has been assessed by Calvo et al. (2021), who conclude that their results should warn of the farming pressure on the organisms and habitats of this Mediterranean ecosystem. Since the designation of this area as a protected park, its management has been characterized by unsustainable dynamics and, most of all, political, institutional, and financial inertia (Jégou and Sanchis Ibor, 2019). It was only in the early 1990s that different engineering solutions were implemented, as described by Martín et al. (2020). In addition, the recent reforms of the Common Agricultural Policy specifically address these agri-environmental systems, providing specific economic support for sustainable farming practices developed within the Natura 2000 network. For example, the burning of straw has been prohibited since 2017, and although these measures are improving the water quality of the L'Albufera lagoon and increasing waterfowl, much remains to be done, even more so given the challenge of climate change, which may increase the salinity of the lake (Martín et al., 2020). Considering the unique characteristics of this environment, it is crucial to find solutions to reduce the impact of rice cultivation in the area.

This study applies to life cycle assessment to estimate the environmental impacts of conventional rice cultivation in L'Albufera (Valencia, Spain). An alternative scenario where the straw is incorporated into the soil, according to carbon farming recommendations, is evaluated to identify potential environmental improvements. The influence of climate and soil characteristics on on-field emissions was considered using the Denitrification-Decomposition (DNDC) model and compared with Tier 1 conventional models. Annual changes in SOC are also considered. The assessment of current practices is the first step to provide recommendations to increase the environmental sustainability of rice production in line with the EU's Farm-to-Fork strategy (European Commission, 2022) and to achieve the Sustainable Development Goals, namely SGD-12 (sustainable production and consumption) and SDG-13 (climate action).

2. Literature review

LCA provides a holistic evaluation of the environmental aspects and potential environmental impacts throughout the life cycle of a product (ISO, 2006). Since its appearance, the method has become important in environmental decision-making, particularly in Europe (Sala et al., 2016). LCA studies on rice have been carried out in different countries, mainly in East Asia, such as Thailand (e.g., Thanawong et al., 2014), Japan (e.g., Hokazono and Hayashi, 2012), Philippines (Nguyen et al., 2019) or Malaysia (Rahman et al., 2019), but also in other countries such as Iran (Khoshnevisan et al., 2014) or USA (Brodt et al., 2014). Existing LCAs assessing European rice production mainly focus on Italy (Bacenetti et al., 2016; Fusi et al., 2014; Zoli et al., 2021). Previous studies by Martínez-Eixarch et al. (2018) and Martínez-Eixarch et al. (2021) have estimated CH_4 and other GHG emissions in the Ebre Delta (Catalonia, NE Spain), an important rice cultivation area in the Mediterranean (Spain). However, to the author's knowledge, a holistic assessment of the environmental impacts of rice from Spain's Mediterranean regions has not been carried out.

Previous LCA studies on rice highlight the relevancy of on-field emissions in many impact categories, such as climate change, acidification, or eutrophication (Bacenetti et al., 2016; Hayashi et al., 2016; Thanawong et al., 2014). This underlines the importance of adequately selecting the methods to estimate these emissions, especially CH_4 and those of reactive N, such as NH_3 or N_2O. However, there is no standard procedure for accounting for on-field emissions in agricultural LCAs (Goglio et al., 2018). Most LCAs rely on the Tier 1 approach of the IPCC guidelines (IPCC, 2006) and the updated refinement (IPCC, 2019), which provide default emission factors that do not differentiate the influence of climate, soil characteristics, fertilizer type, and timing of application, or the irrigation system (Aguilera et al., 2021). Perrin et al. (2014) and Andrade et al. (2021) recommend combining soil N balances with mechanistic and dynamic models to develop generic tools for calculating N-related emissions from agri-food production. These models require many input data (e.g., soil composition, application practices, precipitation, etc.) to be adapted to the region under study. Escobar et al. (2022) used LEACHN (Hutson and Wagenet, 1992) to estimate N emissions from fertilizers in a case study on rice production in the Senegal River Valley and followed the Tier 1 model from the IPCC guidelines (2006, 2019) to calculate CH_4 emissions, as LEACHN does not allow the calculation of this emission. However, it must be noted that no literature has yet been found to validate the use of this model for paddy rice.

As outlined in Section 1, increasing SOC is a major concern. Recently, initiatives, such as the 4 per mille (Minasny et al., 2017) and the EU Soil Strategy 2030 (EC, 2021), have advocated for increasing SOC through alternative soil management practices. In this context, two concepts need to be distinguished (Chenu et al., 2019): carbon sequestration and carbon storage. The former is defined by Olson et al. (2014) as the process by which plant debris (stems, leaves, and roots) and other organic materials transfer CO_2 from the atmosphere to the soil as part of soil organic matter (humus). SOC can be stored in the short term (it is not immediately released into the atmosphere) or in the long term, in which carbon is retained for millennia, resulting in carbon sequestration in the soil (soil carbon stocks). This means that C sequestration is quantified for a given period of time; in particular, following the IPCC recommendation, it is assumed that the rate of change in SOC due to a change in management practices reaches an equilibrium after 20 years. Once this equilibrium is reached, the soil C content is influenced mainly by climate variability, with no clear trend in net C change (Goglio et al., 2015). Chenu et al. (2019) define carbon storage as the increase in SOC stocks over time in the soils of a given land unit, which is not necessarily associated with a net removal of CO_2 from the atmosphere. According to these authors, while long-term carbon storage contributes to climate change mitigation, increasing C stocks positively impacts soils, as labile fractions of SOC are essential for maintaining soil fertility, soil physical conditions, and soil biodiversity. In the context of agricultural LCA, there is currently no consensus on how to account for the effects of changes in SOC in farming systems, either as emissions contributing to climate change or as a separate category that can be used to assess changes in soil quality (Goglio et al., 2015). A standard procedure for accounting for soil C in agricultural LCA is also needed (Bessou et al., 2020; Goglio et al., 2015). In this sense, Joensuu et al. (2021) conclude that the timing should be a deliberate choice of the planning phase of the LCA study, and more explicitly, knowing SOC annual variation can be interesting if it is intended to identify ways to improve carbon sequestration derived from the agricultural management in place at the time.

The DNDC model (EOS, 2012) is a mechanistic model used to evaluate soil C and N dynamics in agricultural, wetland, forest, and grassland ecosystems. Indeed, the use of DNDC to estimate on-field emissions in paddy fields under different management practices and regions has been validated with reliable results (Katayanagi et al., 2017; Khokhar et al., 2022; Oo et al., 2020; Zhang et al., 2022). Yin et al. (2020) also reviewed the research progress of the DNDC model in assessing the impacts of farming practices on SOC and the potential implications for C mitigation in paddy ecosystems. These authors conclude that the model effectively estimates different farming practices' carbon sequestration and emission mitigation potential.

3. Material and methods

3.1. Description of the system under study

L'Albufera de Valencia is a shallow coastal lagoon with an average depth of 1 m, located on the Mediterranean coast to the south of Valencia (Spain). It occupies an area of 23.94 km^2. The climate in this area is classified as warm Mediterranean, characterized by mild temperatures (average annual temperature of 17.8 °C) and occasional rainfall at the equinoxes. The water deficit is clearly defined with an annual rainfall of 522 mm and an evapotranspiration of 856 mm (MTERD, 2021). Soils are classified as hydric, with a saline and carbonated character, with a high organic matter content that increases with depth (Moreno-Ramón et al., 2015).

The plot under study is located in Sueca (Valencia, Spain) and covers an area of 3.7878 ha. The soil of the plot is saline (4.92 dS/m), with a moderately alkaline pH (7.71), a high organic carbon content (43.3 g/kg soil), and a total nitrogen content of 0.31 %. In this farm, the cultivar Bomba (*Oryza sativa* L.) is grown, characterized by a lower nitrogen requirement than other varieties grown in the area and a low yield, with values ranging from 4211 to 4813 kg·ha^{-1} (Franch et al., 2021). This plot was selected because it follows the representative practices of the area. It is located in one of the so-called *tancats*, land reclaimed from the lake, whose surface is a few meters below the water level in the lake. The season begins with a fallow period, from mid-October to February, in which the gates that control the water level in the plots are open, and the water overflows and floods the fields, increasing the area of the lake (winter flooding period); this winter flooding is locally called *perellonà*. During this period, the rice fields are cleared of weeds, weed seeds, fungi, etc. (Osca et al., 2021); moreover, flooded fields are an essential habitat for winter waterfowl. At the end of February, the gates are closed, and the water is pumped out of the fields. In this way, the fields begin to dry out.

Before sowing, the land is worked to create a favorable environment for the plant to germinate. First, a tractor lightly beats the wet soil to aerate it. Later, hooks attached to the tractor help dry the soil. Then, the soil is levelled, and the hooks are used again. The last step before sowing is to fertilize the soil. In particular, in the studied plot, the mineral fertilizer NPK 30-10-12 is applied at 130 kg ha^{-1}. In mid-May, the rice is sown by tractor at a density of 150 kg ha^{-1}, after which the field is flooded again. The total growing cycle is about 120 days.

Water management is a relevant aspect of rice cultivation. Traditionally, the field remains flooded during the cultivation period. An electric pump extracts the water from the field, which stops draining on weekends to maintain the desired water level. In the farm studied, the field is drained five times (the so-called *eixugons*). The first is after the fallow period to prepare the soil; the second, lasting 3–4 days, is in February, ten days after sowing, to allow for field work; the third drainage is in May, also lasting 3–4 days, to allow for the application of phytosanitary products; the fourth drainage is in July to increase the effect of herbicides, and the last drainage is in August, for a week, to dry out the algae that have developed. Once the crop is established, the water level is maintained until two weeks before harvest, around mid-September. Herbicides are used throughout the crop cycle to prevent weed growth, starting one month after sowing, together with fungicides to control *Pyricularia oryzae*. Harvesting is carried out in mid-September with a combine harvester, with a rice yield of 5316 kg of paddy rice ha^{-1} in the year studied. After harvesting, the straw is removed. The straw is used in bedding for cattle, although it has no economic value because it was burned until the recent ban, and there is no market for straw today. Further details on the type and dose of pesticides applied, together with data on machinery use and water management, are shown in Section 3.3 and Table 1.

3.2. Goal and scope definition

This study aims to determine the environmental impact of rice production in a representative plot in L'Albufera (Valencia, Spain), as presented in Section 1. The functional unit (FU) of the study, in which the results are expressed, must reflect the primary function of the system to be studied, which in the case of agricultural systems is to produce food; therefore, 1 kg of paddy rice (14 % moisture content) is the FU chosen. The system boundaries were set "from cradle to farm gate", in other words, all the processes from the production of agricultural inputs were considered, going through all the tasks from the beginning of the cultivation to rice harvesting (Fig. 1). Hence, the processes that rice undergoes once harvested, such as transport to the mill and further processing, were not considered. The system comprises the following stages, namely: a) fertilizer production; b) production of phytosanitary products; c) irrigation, where the energy consumption of the pump and the blue water consumption of the crop are considered; d) on-field emissions, which include N and P emissions caused by the application of fertilizers, the primary distribution of pesticides immediately after their application, and CH_4 emissions from the anaerobic decomposition of soil organic matter; e) use of machinery for on-field operations such as soil preparation, pesticide, and fertilizer application, etc., including the use of a helicopter to spray the fungicides.

Besides the actual farming practices, an additional scenario was evaluated where straw is chopped and incorporated into the soil to account for its influence on soil emissions and SOC. In this scenario, inputs such as water, fertilizer, and pesticides remain unchanged, and the grain yield is unaffected. These assumptions are based on a study by Ribó et al. (2017), who investigated different straw management options without changing the inputs and observed no variations in the yield.

Regarding temporal boundaries, a cropping season corresponding to the year 2020 was considered, starting immediately after the previous season's rice harvest, followed by a fallow period before rice cultivation, and ending with rice harvest. The fallow season is often neglected in these studies; however, according to Martínez-Eixarch et al. (2018),

Table 1
Farming inputs and machinery operations considered in the LCI of rice in *l'Albufera* (Valencia, Spain).

Farming practices	Units	Dose	Application date or operation time (h · ha^{-1})	Machinery
Machinery				
Muddling	L · ha^{-1}	12	1.00	Tractor with caged wheels
First tilling with hooks	L · ha^{-1}	5.75	0.41	Tractor with hooks
Laser levelling	L · ha^{-1}	8	0.8	Tractor with leveller
Second tilling hooks	L · ha^{-1}	5.75	0.41	Tractor with hooks
Fertilizing	L · ha^{-1}	4.5	0.35	Tractor with fertilizer roller
Cover with fertilizer roller	L · ha^{-1}	8.2	0.55	Tractor with roller
Sowing	L · ha^{-1}	4.5	0.35	Tractor with a broadcast seeder
Spraying	L · ha^{-1}	3	1.33	Tractor with sprayer
Helicopter spraying	L · ha^{-1}	200	33.33	Helicopter
Harvesting	L · ha^{-1}	15	0.60	Combine harvester
Fertilizers				
NPK 30-10-12	kg · ha^{-1}	130	10/5/20	–
Herbicides				
Cyhalofop-butyl	L · ha^{-1}	1.5	29/5/20	
penoxsulam + Cyhalofop-butyl	L · ha^{-1}	3	10/6/20	
2,4-D	L · ha^{-1}	1	7/7/20	
Bentazone	kg · ha^{-1}	1.5	7/7/20	
Acetamiprid	kg · ha^{-1}	0.15	7/7/20	
Fungicides	–	–	–	–
Procloraz	L · ha^{-1}	1	7/7/20	
Azoxystrobin + Difenoconazole	L · ha^{-1}	1	27/7/20	
Azoxystrobin + Difenoconazol	L · ha^{-1}	1	12/8/20	
Watering				
Water dose	L · ha^{-1}	n.a.	n.a.	n.a.
Pumping[a]	kWh · ha^{-1}	56,849.92	n.a.	Water pump

n.a.: data not available.
[a] To drain water from the field.

straw and water management during the fallow season are crucial for accurate estimation of CH_4 emissions. Rice cultivation also produces straw, which is removed and has no economic value; therefore, no allocation was carried out, and all the environmental burdens associated with rice cultivation were attributed to the rice grains. Similarly, no allocation was carried out in the scenario where the straw is chopped and incorporated into the soil.

3.3. Inventory analysis

The Life Cycle Inventory (LCI) aims to identify and quantify all inputs (consumption of resources and materials) and outputs (emissions to air, soil, water, and waste generation) related to the system under study. To achieve this, primary and secondary data were used. Primary data sources are first-hand information specific to the activity in question. In this case, the information was obtained from the surveyed producer on the practices followed in 2020 (Table 1). Secondary data obtained from commercial LCI databases were used to model the background system, namely Ecoinvent 3.8 (Wernet et al., 2016) and GaBi DB v10 (Sphera Solutions GmbH, Leinfelden-Echterdingen, Germany). The secondary data and the models used to estimate on-field emissions are explained below.

Production of fertilizers and plant protection products. The resources consumed and the emissions generated in the manufacturing of the fertilizer NPK 30-10-12 were simulated using the production of generic nitrogen, phosphorus, and potassium fertilizers from the Ecoinvent 3.8 database. The active ingredients of the commercial plant protection products used were not found in the available databases, so the generic pesticide production process from Ecoinvent 3.8 was used.

Field operations. A tractor is used for field operations (tillage, fertilization, seeding, and herbicide application), except for fungicide application (azoxystrobin and difenoconazole), sprayed with a helicopter, and harvesting, made with a combine harvester. The emissions derived from the tractor and the combine harvester were calculated by adapting the respective processes available in GaBi DB v10 (Sphera, Chicago, USA) according to the rice producer respondent's fuel consumption and operating time. The emissions from the helicopter were estimated using the Ecoinvent 3.8 process, considering that 100 ha are sprayed in 2.75 h.

Water use and water consumption. As described in Section 3.2, the plot is irrigated directly from *l'Albufera* Lake by opening the tancats' floodgates, implying that no energy is consumed. However, an electric pumping system allows a constant and continuous flow of water out of the field (water flow of 3000 L min^{-1}) to maintain the level of the water sheet (7–10 cm in the initial period and 10–15 cm in the rest). As also explained in Section 3.2, five drains are carried out, and each drain requires the pump to run for approximately 10, 12, 14, and 14 h, respectively, since the first drain after the fallow period does not need the pump. To determine the energy demand of the pump and its emissions, the irrigation pump process of GaBi DB v10 was adapted in terms of flow rate and the pumping time, using the process corresponding to the production of the Spanish electricity mix from the same database. It must be noted that the operation of the pump to maintain the water level when the field is flooded was not taken into account, as reliable data was not available.

In a LCA, blue water consumption refers to the water withdrawn from a watershed and not returned to it. It is, therefore, caused by evaporation, transpiration, addition to a product, or discharge to another watershed or the sea (Hoekstra et al., 2011). In this case study, blue water consumption was determined by considering the daily evapotranspiration of the crop (ETc) and the effective precipitation (*Pe*). ETc is the combination of two independent processes by which water is lost: evaporation at the soil surface and crop transpiration; it was calculated as (FAO, 1998):

$$ET_C\left(mm \cdot day^{-1}\right) = K_C * ET_0\left(mm \cdot day^{-1}\right) \quad (1)$$

where K_c is the crop coefficient (dimensionless), namely 1.05 in the initial growth stage (first 30 days), 1.1 in the development stage (60 days) and 1.15 in the final stage (last 30 days). ET_0 is the reference evapotranspiration estimated with the FAO Penman-Monteith method, obtained from a weather station in Polinyà de Xúquer, the closest town to *l'Albufera* for which climatic data was available. The total ET_0 values for the rice growth stages mentioned above are 5.197 mm, 5.073 mm, and 5.045 mm, respectively (SIAR, 2021). The ETc was calculated with these data, resulting in 5.46, 5.56, and 5.58 mm·day^{-1}·ha^{-1} in the initial growth, development, and final stages, respectively.

Pe is the fraction of rain stored in the root zone that plants can effectively use (FAO, 1986). *Pe* (mm) was calculated using the USDA SCS method, which varies with total monthly precipitation (*Pt*, mm) (FAO, 1998):

$$Pe = Pt*(125 - 0.2*P_t/125) \text{ if } Pt < 250\ mm \quad (2)$$

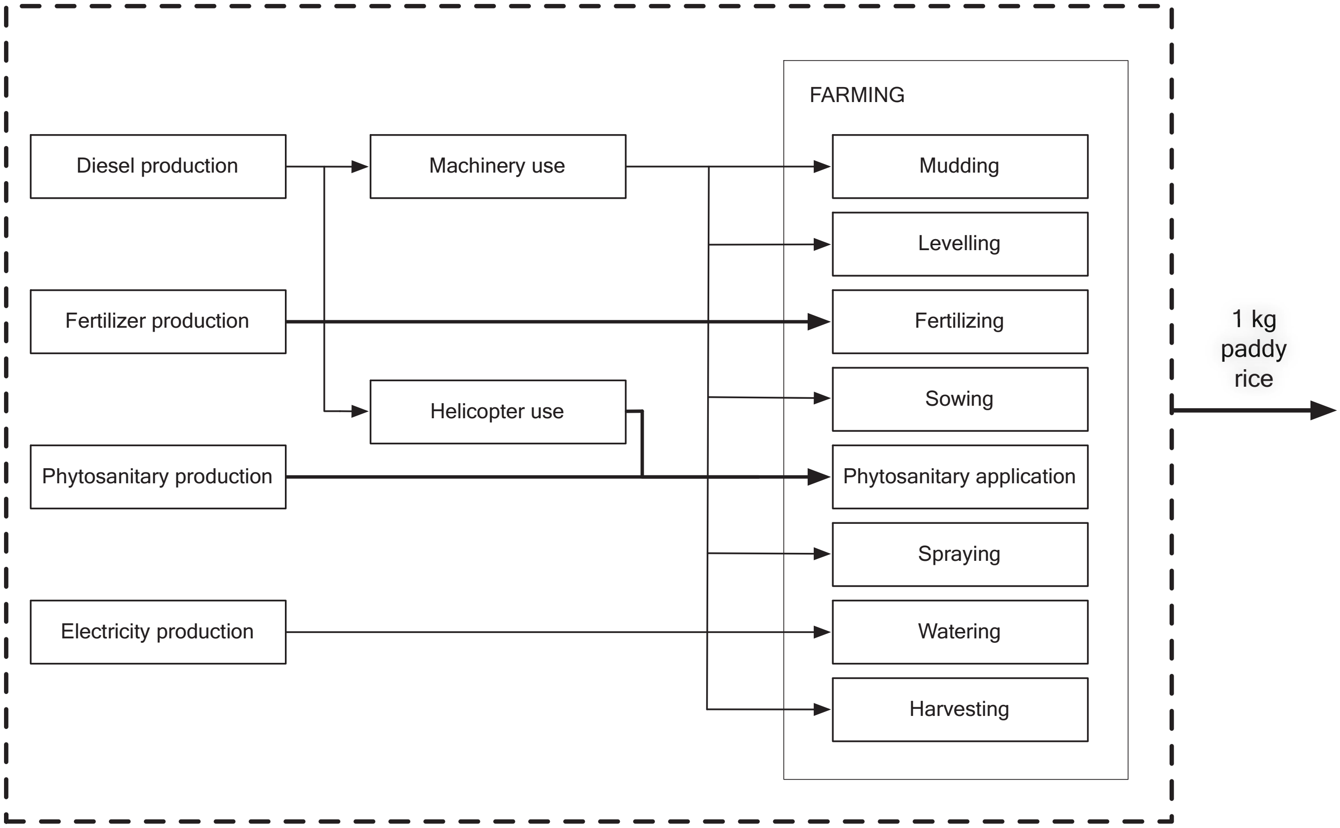

Fig. 1. System boundaries considered for paddy rice production in *L'Albufera* (Spain).

$$Pe = 125 + 0.1 * P_t \ if \ Pt > 250 \ mm \tag{3}$$

The amount of total blue water consumed by evapotranspiration was calculated by multiplying the evapotranspiration of the crop in each growing season by the duration (days) of each growing period of the crop and subtracting the effective precipitation if it is less than the evapotranspiration; otherwise, the water consumption is zero. The total water use was 318.24 mm·ha^{-1}.

On-field field emissions. On-field emissions were calculated using a Tier 3 model, the DNDC developed by Li et al. (1992a, 1992b), and later improved to fit rice cultivation areas accurately (Fumoto et al., 2008). This model simulates the biochemical cycling of carbon and nitrogen in agricultural ecosystems and estimates CH_4, N_2O, NO, and NH_3 emissions and the SOC variation during the assessment year. As commented in Section 1, the model has been used extensively in paddy studies and provides reliable results. The model requires data specific to the study area, including climatic and edaphic data, and the management data specified in previous sections. It was previously calibrated in a climatically normal year, which means that the climatic parameters of that year are within the average values of a series of measurements of >30 years. For this purpose, gas samples were taken from four plots in L'Albufera using three sealed methacrylate chambers per plot and analyzed by gas chromatography. During a farming season, the values sampled and analyzed in situ (edaphic variables, crop management, climatic data) were included, and some initial parameters predefined by the model were modified to fit the DNDC (microbial activity index, rice variety, root and stubble incorporation ratio, and organic matter decomposition ratio). The correlation coefficient between observed and simulated values (0.98) and the root mean square error (RMSE of 0.045) were calculated to validate the adjusted model. The normalized RMSE value was 23.8 %, indicating moderate agreement between the experimental data and the model (Yang et al., 2014). The model fits with errors <10 %, precisely 1.31 % in the case of straw incorporation (128.8 vs. 127.1 kg CH4 · ha^{-1}·year^{-1} in the measured data and DNDC estimates, respectively) and 7.64 % in the case of straw removal (104.8 vs. 113 kg · ha^{-1}· year^{-1}).

The DNDC does not estimate PO_4^{3-} leaching to groundwater and runoff to surface water, and the guidelines used in the Júcar Hydrographic Confederation (CEDEX, 2017) were thus followed since the studied plot is in the Júcar watershed. According to these guidelines, the final losses of this compound are calculated taking into account the balance between the inputs (flooding conditions, precipitation, filtrations) and outputs (drainage, crop, and soil-water interactions), as well as the phosphorus concentration in water (surface and groundwater), and assuming that the phosphorus contributions from the rain are zero.

Emissions from the DNDC model were compared with those estimated using Tier 1 and Tier 2 emission factors. Direct and indirect N_2O emissions to air and NO_3^- emissions to freshwater were calculated using the Tier 1 approach of the IPCC (2006) and the recent refinement (IPCC, 2019). Tier 2 emission factors from EMEP/EEA (Hutching et al., 2019) were used to estimate NH_3 and NO_x emissions to air. CH_4 emissions to air were calculated following IPCC (2006, 2019), considering the time factor to account for the water regime during both the pre-season and the cultivation period and that no organic amendments were applied. In the scenario with straw incorporation, the amount of straw generated and its N content were taken from Ribó et al. (2017). Leaching and runoff of soluble PO_4^{3-} were estimated following Nemecek et al. (2014).

Pesticide emissions. The primary distribution of pesticide products to the environmental compartments was estimated using the Pest LCI Consensus (Fantke et al., 2017a, 2017b). This model calculates both on-field and off-field emissions of the active compound of each plant protection product in the three environmental compartments (air, water, and agricultural soil). Off-field pesticide emissions are further distributed among the environmental compartments (agricultural soil, freshwater, and other soil).

3.4. Life cycle impact assessment

The Life Cycle Impact Assessment relates the inventory emissions and resource use to potential impacts on the environment, human health, and resource availability to classify, characterize, and assess the significance of the potential impacts. The Environmental Footprint (EF) v3.0 methodology (Damiani et al., 2022) was used to describe these impact categories. Blue water consumption was assessed using the AWARE 1.2C (Boulay et al., 2018) characterization factors for the global average for unspecified water except for the irrigation stage, where those for farming in Spain were used.

4. Results

4.1. On-field emissions

The results of the on-field emissions for current practices (straw removed) estimated with the two approaches are shown in Table 2. When the DNDC model is used to calculate the N balance, the total net loss of soil N is mainly determined by the volatilization of NH_3 (41.29 kg $NH_3 \cdot ha^{-1}$). In contrast, the fluxes of N_2O, NO, and NO_3^- are orders of magnitude lower (3.46 kg $N_2O \cdot ha^{-1}$, 1.29 kg NO ha^{-1} and 0.4 kg $NO_3^- \cdot ha^{-1}$ respectively). When comparing these emissions with those estimated by the Tier 1 and Tier 2 approaches, the most notable differences are observed in the NO_3^- values, which are >90 times higher when using the IPCC emission factor. Nitrate is easily denitrified to N_2O or N_2 under anaerobic conditions, and in addition, nitrate N can alter the redox potential of soils, reducing CH_4 production (Patrick and Mahapatra, 1968; Wang et al., 2020). NH_3 emissions estimated with DNDC are more than ten times higher than those calculated with the Tier 2 EMEP/EEA method, while small differences are observed for the remaining N emissions. Methane emissions also show different values depending on the approach used. The difference in C emissions between the DNDC and the IPCC model lies in the accuracy of the information that can be recorded in the case of the DNDC since it considers the C losses that occur when the field is flooded (CH_4) and drained (CO_2). The DNDC model estimates 108.1 kg CH_4 ha^{-1}, about 40 % less than when using the IPCC Tier 1 model (2006, 2019). In rice fields, anaerobic soil conditions during the flooding period cause an increase in CH_4 emissions (due to the activity of anaerobic soil bacteria) and a decrease in CO_2 emissions (limited activity of aerobic bacteria). This effect is the opposite of what happens when the field is drained, where there is an increase in CO_2 and N_2O emissions. The net SOC balance inherent in these situations reveals the capacity of the soil to store carbon as a function of the wetland or, conversely, to generate a loss in the soil due to farming management. Moreover, if we add to this fact the work done in the fields (tillage, plowing and levelling), during the dry period of the fields, greater CO_2 emissions are generated and, therefore, a loss of carbon in the soil, resulting in a carbon deficit in a crop year according to the DNDC model (Table 2).

Table 2
On-field emissions per ha and year estimated with the DNDC Tier 3 model versus the emissions estimated with IPCC (2019) Tier 1 and EMEP/EEA (Hutching et al., 2019) Tier 2.

	Straw removed		Straw incorporated	
Emissions ($kg \cdot ha^{-1} \cdot year^{-1}$)	DNDC	IPCC/EMEP	DNDC	IPCC/EMEP
NO_3^-	0.4	41.0	0.4	73.4
NH_3	41.9	3.7	36.6	3.9
N_2O	1.4	0.2	1.3	0.8
NOx	0.2	1.6	0.2	2.8
SOC*	−1984	n.e.	−876	n.e.
CH_4	108.1	248	131.1	13,155
PO_4^{3-}	5.0	1.5	5.0	1.5

n.e.: data not estimated; SOC*: annual variation of Soil Organic Carbon.

From the results of the P balance, the total losses due to runoff and leaching reached 0.82 kg P $\cdot ha^{-1}$, that is, 10.5 % of the total phosphorus introduced into the paddy fields through irrigation water, fertilization, and groundwater contributions. The value calculated following the recommendations of the World Food Database (Nemecek et al., 2014) is about three times lower, 0.25 kg ha^{-1}; this lower value can be attributed to the method that only considers P from fertilization.

The primary distribution of the active compounds of the plant protection products calculated with PestLCI Consensus is shown in Table 3. The greatest fraction of the active compounds reaches the crop and the agricultural soil (about 80–95 % of the total active compound), whereas the amount transported in the air is about 5 %. This fraction increases to 12.5 % in the case of fungicides, as they are applied by helicopter. Regarding freshwater, only 0.03 % of the active compounds used reach this environmental compartment.

4.2. Environmental impacts of current practices

The environmental impacts of the current practices, without incorporating the straw into the soil, were calculated using the on-field emissions estimated with the DNDC model. The scores obtained per mass of rice for all impact categories are shown in Table 4. On-field emissions associated with the farming practice is the stage that mainly contributes to most of the environmental impacts (Fig. 2); as for eutrophication, it represents 100 % of marine eutrophication due to N emissions, 94 % of freshwater eutrophication, mainly due to PO_4^{3-}leaching, and 97 % of terrestrial eutrophication. It also accounts for >100 % of the toxicity impacts, namely freshwater ecotoxicity and human toxicity, both carcinogenic and non-carcinogenic, due to pesticide emissions. The great share of the on-field emissions stage in the acidification category (96.9 %) is due to NH_3, which, together with NO_x emissions, is also responsible for particulate matter formation (94 %). On-field emissions also represent 82 % of the total climate change impact, mainly due to N_2O from fertilizer application and CH_4 from the anaerobic decomposition of organic matter when the field is flooded. For the remaining environmental impact categories, the production of plant protection products leads to ozone depletion (81 %), caused by the release of halogenated organic compounds, and metal depletion (73 %). Agricultural machinery is responsible for 53 % of the photochemical ozone formation, and this stage and the fertilizer production represent the highest share of land use (36 % each). The electricity needed to pump water from the fields is the main cause of ionizing radiation (78 %). Finally, the use of fossil resources is mainly caused by the production of fertilizers (40 %), followed by the use of machinery and the drainage pump (ca. 23 %). It must be noted that the agricultural machinery stage includes the use of helicopters for fungicide spreading; however, when considered alone, it does not make a remarkable contribution to any of the impact categories studied (between 0.01 and 1.29 %, depending on the impact).

4.3. Environmental effects of incorporating the straw into the soil

Incorporating the straw into the soil affected on-field emissions, as shown in Table 4 for the two approaches. When estimating the emissions using the DNDC model, the main differences concern C emissions. CH_4 emissions increased by 21 % due to SOC variation from straw incorporation. Additionally, CO_2 emissions increased from 4136.2 kg C/ ha to 5430.9 kg C/ha; consequently, net emissions reveal a >50 % decrease in SOC. However, results can change when the effect of straw incorporation is reproduced over time. For example, a 5-year simulation with the calibrated model and the same conditions shows a tendency to accumulate carbon in the soil and a 49 % increase in SOC compared to the 1-year simulation shown in Table 2. However, CH_4 emissions were 1.3 times higher than the initial year. Pampolino et al. (2008) did not find changes in SOC under a 50-year long-term experiment in the Philippines. In contrast, Alberto et al. (2015) concluded from a study in

Table 3
Primary distribution of the active compounds of the plant protection products applied to the crops (kg ha^{-1}) into the environmental compartments.

Active compound	Air	Soil	Crop	Other agricultural soil	Freshwater	Other soils
Cyhalofop butyl	$4.20 \cdot 10^{-2}$	$6.37 \cdot 10^{-1}$	$1.58 \cdot 10^{-1}$	$1.04 \cdot 10^{-3}$	$2.76 \cdot 10^{-5}$	$1.69 \cdot 10^{-3}$
Penoxsulam	$2.42 \cdot 10^{-1}$	3.66	$9.11 \cdot 10^{-1}$	$6.01 \cdot 10^{-3}$	$1.58 \cdot 10^{-4}$	$9.72 \cdot 10^{-3}$
2,4-D	$2.50 \cdot 10^{-2}$	$2.38 \cdot 10^{-1}$	$2.36 \cdot 10^{-1}$	$5.15 \cdot 10^{-4}$	$1.36 \cdot 10^{-5}$	$8.35 \cdot 10^{-4}$
Bentazona	$6.53 \cdot 10^{-2}$	$6.20 \cdot 10^{-1}$	$6.16 \cdot 10^{-1}$	$1.62 \cdot 10^{-3}$	$4.29 \cdot 10^{-5}$	$2.63 \cdot 10^{-3}$
Acetamiprid	$1.50 \cdot 10^{-3}$	$1.43 \cdot 10^{-2}$	$1.42 \cdot 10^{-2}$	$3.73 \cdot 10^{-5}$	$9.87 \cdot 10^{-7}$	$6.04 \cdot 10^{-5}$
Procloraz	$2.25 \cdot 10^{-2}$	$2.14 \cdot 10^{-1}$	$2.12 \cdot 10^{-1}$	$5.59 \cdot 10^{-4}$	$1.48 \cdot 10^{-5}$	$9.06 \cdot 10^{-4}$
Azoxistrobin	$5.00 \cdot 10^{-2}$	$1.42 \cdot 10^{-1}$	$1.96 \cdot 10^{-1}$	$4.57 \cdot 10^{-3}$	$1.21 \cdot 10^{-3}$	$7.39 \cdot 10^{-3}$
Difenoconazol	$3.13 \cdot 10^{-2}$	$8.90 \cdot 10^{-2}$	$1.22 \cdot 10^{-1}$	$2.85 \cdot 10^{-3}$	$7.55 \cdot 10^{-5}$	$4.62 \cdot 10^{-3}$

Table 4
Environmental impact results per 1 kg of paddy rice produced in L'Albufera (Valencia, Spain) following current practices, where straw is taken from the field, and in an alternative scenario, straw is incorporated into the soil.

	Straw removed	Straw incorporated
Acidification (Mole of H^+ eq.)	$2.46 \cdot 10^{-2}$	$2.16 \cdot 10^{-2}$
Climate Change, total (kg CO_2 eq.)	$7.56 \cdot 10^{-1}$	$8.68 \cdot 10^{-1}$
Ecotoxicity, freshwater (CTUe)	$4.62 \cdot 10^{3}$	$4.62 \cdot 10^{3}$
Eutrophication, freshwater (kg P eq.)	$3.29 \cdot 10^{-4}$	$3.29 \cdot 10^{-4}$
Eutrophication, marine (kg N eq.)	$4.81 \cdot 10^{2}$	$4.81 \cdot 10^{2}$
Eutrophication, terrestrial (Mole of N eq.)	$1.10 \cdot 10^{-1}$	$9.63 \cdot 10^{-2}$
Human toxicity, cancer (CTUh)	$3.01 \cdot 10^{-8}$	$3.01 \cdot 10^{-8}$
Human toxicity, non-cancer (CTUh)	$1.48 \cdot 10^{-6}$	$1.48 \cdot 10^{-6}$
Ionizing radiation, human health (kBq U235 eq.)	$2.73 \cdot 10^{-2}$	$2.73 \cdot 10^{-2}$
Land Use (Pt)	$8.53 \cdot 10^{-1}$	$8.53 \cdot 10^{-1}$
Ozone depletion (kg CFC-11 eq.)	$3.56 \cdot 10^{-8}$	$3.56 \cdot 10^{-8}$
Particulate matter (Disease incidences)	$1.77 \cdot 10^{-7}$	$1.56 \cdot 10^{-7}$
Photochemical ozone formation, human health (kg NMVOC eq.)	$9.92 \cdot 10^{-4}$	$1.04 \cdot 10^{-3}$
Resource use, fossils (MJ)	1.90	1.90
Resource use, mineral and metals (kg Sb eq.)	$4.08 \cdot 10^{-6}$	$4.08 \cdot 10^{-6}$
Water use (m^3 world equiv.)	$5.93 \cdot 10^{-2}$	$5.93 \cdot 10^{-2}$

the same country that continuous straw incorporation increases SOC due to the slower decomposition of organic matter. Therefore, straw residues, flooding conditions, application time, and tillage are directly related to SOC accumulation and gas emissions (Chivenge et al., 2020).

For N emissions, differences were observed for NH_3 and N_2O, which decreased by 13 % and 11 %, respectively, compared to those estimated when the straw was baled and removed from the field. Studies by Zhang et al. (2010) and Wang et al. (2015) also concluded that the addition of straw reduces N emissions. The application of straw inevitably leads to an increase in SOC, which increases the demand for N by soil microbes as field temperatures rise. This increase in N consumption by the microbial mass causes a decrease in N_2O in the soil, limiting its presence and reducing N_2O emissions in general. Instead, NO_3^- emissions to water and NO_x emissions to air remain unchanged because soil management (flooding/oxidizing conditions) and soil N availability generally decrease N content. The conventional IPCC (2006, 2019) approach increased all emissions as straw increases SOC and N in soil. CH_4 emissions were 53 times greater than those estimated using the same method but without considering straw incorporation. The primary distribution of the active compounds of the plant protection products remains the same (Table 3), as the doses applied do not change.

The scores of the environmental impact assessment of the alternative scenario where the straw is incorporated into the soil are presented in Table 4, together with the current practice of removing the straw from the field. It must be borne in mind that these impacts have been calculated using the emissions estimated with DNDC and assuming no differences in the yield, as explained in Section 3.2. Therefore, the changes in on-field emissions mentioned above can help to interpret the differences in the impact scores. The 13 % decrease in acidification is related to the same percentage decrease in estimated NH_3 emissions, which also explains the lower scores for terrestrial eutrophication and particulate matter. The increase in the photochemical ozone formation score is mainly due to increased CH_4 emissions. Regarding climate change, although CH_4 emissions increased by 21 %, NO_2 and SOC decreased by 11 % and 52 %, respectively, which explains the 15 % increase in this impact score. The contribution analysis is very similar to that described in Section 4.2, as the difference in the impact scores is moderate or even null, depending on the impact category. On-field emissions are again the main contributor to marine eutrophication (100 %), freshwater eutrophication (94 %), terrestrial eutrophication (96.5 %), and toxicity impacts (ca. 100 %). Changes in CH_4 and N_2O emissions slightly increase the contribution of climate change impacts from 82 % to 84 %. The increase in CH_4 emissions justifies that the contribution of on-field emissions to photochemical ozone formation increases from 25 % to 28 %, and that of machinery use decreases from 53 % to 51 %.

5. Discussion

5.1. On-field emissions and changes in SOC

The results show that on-field emissions from fertilizers and pesticides are crucial for the environmental impact of rice; therefore, the approaches used to estimate these emissions are decisive as remarkable differences in their values are found depending on the approach used (see Sections 4.1 and 4.3). The higher N_2O and NH_3 emissions estimated with DNDC compared to those calculated with the IPCC Tier 1 method are attributed to the fact that the DNDC model considers the effects of climate and soil conditions and management practices, whereas Tier 1 estimates N_2O emissions as a specific percentage of nitrogen inputs. Fitting the model to local management practices is crucial to increase its reliability, and the availability of real crop management data is a critical factor. The phenological cycle of L'Albufera of Valencia (Mediterranean region) shows similarities in the developmental stages in time between dry and wet periods with areas in other parts of the world, such as Vietnam or the United States (Chen et al., 2016; Wei et al., 2021; Vera-Herrera et al., 2021). The DNDC model has been used and fitted in multiple rice case studies, being validated over short time periods, or in simulations with extensive temporal data (Li, 2000; Jagadeesh et al., 2006; Katayanagi et al., 2013; Katayanagi et al., 2016; Shaukat et al., 2022). The models readjusted by the different authors predict greenhouse gas emissions much better than the standard DNDC model. The uncertainties that cause problems in the application of the model are the use of data from large areas and the use of non-localized soil or management data and rice variety specificity (Katayanagi et al., 2016; Smakgahn et al., 2017).

Bateman and Baggs (2005) remarked that the denitrification process (anaerobic) favors N_2O production compared to the nitrification process (aerobic). Therefore, to obtain reliable results, it is critical to consider the type of flooding regime in the paddy field when modelling this emission. Similarly, Yang et al. (2013) concluded that the soil saturation by a water table close to the surface reduced N_2O emissions by >50 % compared to hydric soils under oxidizing conditions (water table far from the soil surface). However, other studies comparing different model applications show disparate results, depending on the information included in the model and the scale of the studies. Li et al. (2001)

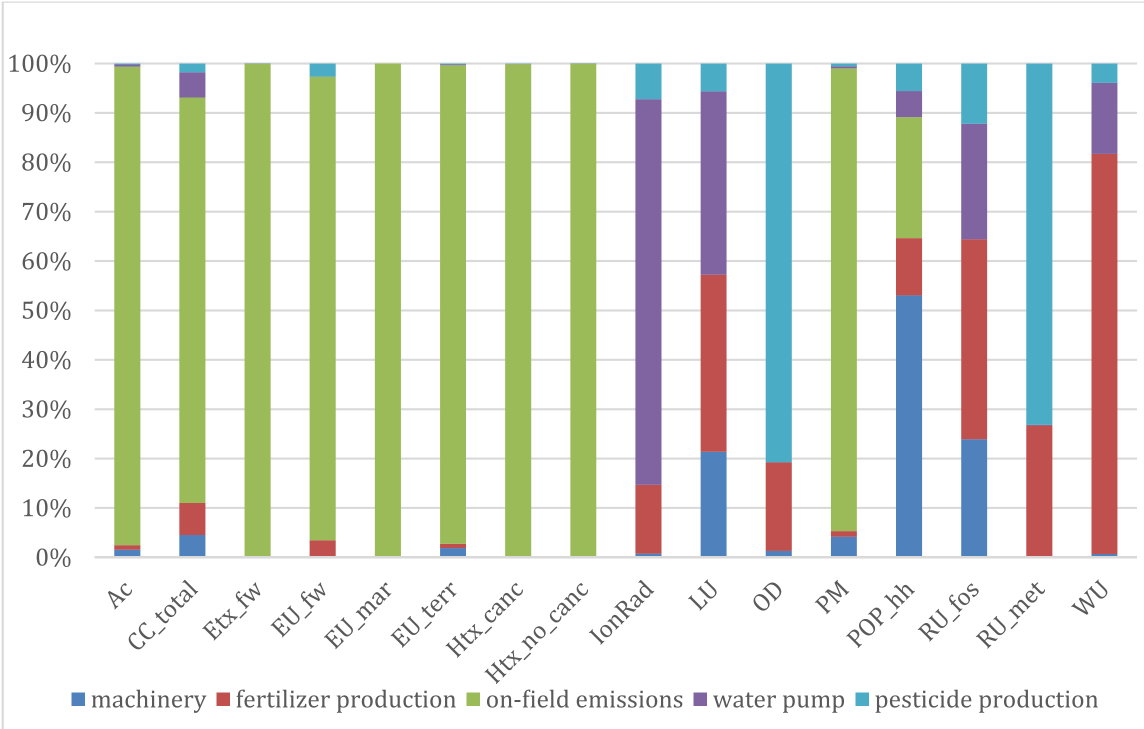

Fig. 2. Contribution of the life cycle stages to the environmental footprint of 1 kg of paddy rice produced following current practices in L'Albufera (Valencia, Spain). Ac: acidification; CC_total: climate change, total; Etx_fw: freshwater ecotoxicity; Eu_fw: freshwater eutrophication; EU_mar: marine eutrophication; EU_terr: terrestrial eutrophication; Htx_canc: carcinogenic human toxicity, recommended and interim; Htx-no_canc: non-carcinogenic human toxicity, recommended and interim; IR: Ionizing radiation, human health; LU: Land use; OD: ozone depletion; PM: particulate matter; POP_hh: photochemical ozone formation, human health; RU_fos: resource use, fossils; RU_met: resource use, mineral and metals; WU: water use.

registered that the DNDC N_2O emissions were lower than the IPCC estimates in the main agricultural region of China, while in the northern and coastal areas of the country, the N_2O estimate was higher than the IPCC outcomes. Shi et al. (2020) applied the DNDC model to a rice area in the Yangtze River (China). They validated the results with two datasets (IPCC and Emission Database for Global Atmospheric Research - EDGAR). They concluded that the DNDC N_2O estimates were comparable to the EDGAR values but one-third of the IPCC results.

The difference in carbon emissions between the DNDC and Tier 1 models lies in the accuracy of the information that can be recorded to fit the DNDC model. DNDC allows the user to define the days and hours in a flood regime accurately, reducing soil or oxidizing conditions due to fluctuations in the water table and surface water layer. Several studies have shown the effect of the water table on field emissions. Specifically, its level in the soil matrix generates anaerobic or aerobic conditions that regulate the decomposition rate of the soil organic matter. Yang et al. (2013) demonstrated that water tables close to the surface generated higher CH_4 emissions (26.9 mg m^{-2} h^{-1}) compared to the average value registered in a deeper water table position (5.46 mg m^{-2} h^{-1}). However, the soils of L'Albufera, close to the Mediterranean Sea, exhibit another characteristic to be considered when estimating CH_4 emissions: soil salinity. Salinity can increase microbial respiration or decrease soil respiration, leading to either a decrease or an accumulation of SOC, depending on the short- or long-term salinity affection (Stagg et al., 2017; Wen et al., 2019). Zhao et al. (2020) combined water table level and salinity conditions in a coastal wetland in China and observed that lower water table level increased CO_2 and decreased CH_4 emissions in coastal wetlands with saline conditions (salt marsh). In the present case study, the effect of rice field management in a specific year was investigated, and the soil salinity data were collected after analyzing soil samples. However, this simulation should be readjusted again for longer-term studies, including the effect of soil salinity. Future studies should also incorporate the potential effect of rising sea levels due to climate change since soil salinity affects carbon storage dynamics.

For P emissions, Leon and Kohyama (2017) calculated phosphorus losses in a paddy field in Japan using a site-specific approach (TP losses of 0.9 kg ha^{-1}) or a non-site-specific approach (5.6 kg ha^{-1}). Straw residue management, fertilizer incorporation, and soil properties were the differences between both methods, concluding that phosphorus losses are more precise when the specific conditions of the study area are considered.

Traditionally, straw has been burned in L'Albufera despite its adverse effects on soil quality, SOC sequestration, and air quality. Burning was preferred by farmers mainly due to its lower cost, reduced weed and disease transmission, and convenience for tillage (Allen et al., 2020). However, as mentioned in Section 1, this practice is now forbidden, except in special situations decided by the local authorities (e.g., pests or problems with straw management), in which burning is allowed. Furthermore, straw management is completely erratic in each plot. Farmers leave the straw in the field depending on whether they are allowed to burn it or remove it (a period of rain after harvesting causes the straw to be left in the field because the machinery cannot remove it). For this reason, straw withdrawal (promoted by the regional government) and incorporation into the soil were simulated. Proposing longer-term simulations would show high variability in terms of SOC variation, as observed in Section 4.3. Hoang et al. (2019) estimated that burning reduced CH_4 emissions by 18–34 % and N_2O emissions by 21–32 % in Vietnam paddy fields. Returning rice straw to the soil can increase SOC because nearly half of the carbon in rice plant residues is in the straw and stubble. However, it should be noted that the carbon from the roots contributes the majority of SOC (Allen et al., 2020). In addition, a positive effect on SOC is observed as it is lower when straw is incorporated, which offsets the more significant impact on climate change, although this effect changes when more extended periods are simulated.

When using DNDC to estimate on-field emissions, the impact scores in case the straw is incorporated into the soil immediately after the harvest showed to be beneficial in some impact categories, namely acidification, terrestrial eutrophication, and particulate matter formation. In contrast, a detrimental effect is observed in photochemical ozone formation and climate change. However, when the impacts are estimated with the conventional IPCC (2006, 2019) method, all the scores increase because, as commented in Section 4.3, all the emissions increase, too. These increases are more marked for climate change (58 % higher with straw inclusion), marine eutrophication (80 %), and photochemical ozone formation (39 %), as the changes in the dominant on-field emissions are of the same order.

5.2. Comparison with other LCA studies and recommendations to decrease the impacts

The comparison of the results of this LCA with selected case studies from the literature in the Mediterranean region is not straightforward, as the assumptions made differ depending on the case study, mainly regarding the system boundaries, the models used to estimate on-field emissions, and the impact assessment methods. In particular, all the reviewed studies applied the Tier 1 approach to estimate CH_4 and N_2O emissions. Fusi et al. (2014) assessed rice IPCC production in northern Italy. They observed that when straw is collected for sale, all the impact scores decrease except for freshwater eutrophication. This can be explained by the fact that as the straw is sold, the environmental loads are allocated between the paddy rice and the straw, and because CH_4 emissions are substantially reduced when using the IPCC (2006) method. On the other hand, the higher score for eutrophication is related to the higher rate of mineral fertilization to compensate for that provided by the straw. Bacenetti et al. (2016) evaluated organic rice in the same Italian region and reached similar results concerning the straw. It must be noted that Bacenetti et al. (2016) compared their impact scores with those of conventional rice from Fusi et al. (2014), which are lower than those of organic rice due to the lower yield and show that the emissions from compost production significantly affect impact scores such as climate change.

Recent studies by Vaglia et al. (2022) and Zoli et al. (2021) assess the environmental impact of alternative management practices in northern Italy and can be used as a guide to reduce the effects of Mediterranean rice from L'Albufera. In particular, Zoli et al. (2021) evaluated the environmental benefits of adopting an alternative water management characterized by an additional aeration period for two rice varieties (Carnaroli and Caravaggio). Their results show that contrary to other studies, alternative water management does not influence grain yield and decreases CH_4 emissions, reducing the impact of climate change. Maris et al. (2016) observed the same trend in an experience developed in a Mediterranean wetland about 200 km north of L'Albufera; CH_4 emissions measured in paddy fields under continuous flood management were significantly higher than under aeration-flooding cycles. L'Albufera of Valencia can be considered a coastal wetland of high ecological richness associated with rice cultivation, which also serves as an example of the complex water management practices found in other Mediterranean wetlands (Jégou and Sanchis Ibor, 2019; Vera-Herrera et al., 2021). Zoli et al. (2021) also analyzed the influence of the adopted water regime on the heavy metal content (namely cadmium and arsenic) of rice samples, observing that the arsenic content in the grain decreased in all the alternative scenarios. In contrast, the cadmium content increased while remaining well below the legal limits. The study by Vaglia et al. (2022) delves deeper into organic rice farming in northern Italy, examining different management practices. Their results showed that nowadays, farmers could reach acceptable yield values by incorporating effective management practices, including the stale seedbed in the dry field, mechanical control of weeds, extending soil aeration, and less input such as green or organic manure. Fertilization raises serious environmental concerns due to the impact of fertilizer production processes and the emissions of nitrogen and phosphorus compounds into the environmental compartments. Along these lines, precision agriculture is a promising solution to reduce these impacts by making application rates more precise and increasing yields (Bacenetti et al., 2020). Field experiments showed that adding biochar to rice soils decreases CH_4 and N_2O emissions (Mohammadi et al., 2016). However, according to Xie et al. (2013), it depends on the soil type, as they observed an environmentally favorable impact when biochar was added to acidic Ultisol but not to alkaline Inceptisol. These authors also measured a significant increase in SOC. Unlike other types of carbon inputs, biochar additions do not substantially increase microbial activity, and SOC is maintained even when ceasing this measure (Paul et al., 2023). Proposing alternatives to mitigate the environmental impacts of agricultural production can be controversial, as sometimes a practice that is beneficial for some impact categories is detrimental for others. For instance, regarding water management, although drying the field during the winter fallow reduces CH_4 emissions, Pérez-Méndez et al. (2022) also observed a significant reduction (between 57 and 75 %) in waterfowl diversity concerning flooded fields, hindering biodiversity conservation.

6. Conclusions

The results of this LCA of rice production in L'Albufera (Valencia, Spain) highlight the role of on-field emissions in the environmental impact scores of these systems. Consequently, the relevance of harmonizing and improving the modelling of these emissions is also highlighted, especially those of NO_3^- and CH_4, which exhibited greater differences depending on the approach used, namely conventional Tier 1 and Tier 2 emission factors and the mechanistic DNDC model. The values obtained by the DNDC model represent a very good fit for the rice conditions in a Mediterranean area if specific management data and environmental characteristics are available. In addition, SOC variation was also estimated because, besides the relevance of this parameter to represent soil quality, it also reflects the carbon sequestration associated with farming practices in line with the European policy on carbon.

Water and straw management are critical to reducing the environmental impact of paddy fields. In the study area, fields are dried several times during the cropping season for technical purposes (e.g., pesticide application), which also benefits on-field emissions. In addition, fields are flooded during the fallow season, which poses an environmental trade-off because, despite the negative effect on CH_4 emissions, it is beneficial for biodiversity, as noted in recent literature. This is positive in protected ecosystems such as L'Albufera, internationally recognized for its birdlife, and other areas in the Mediterranean basin. Straw burning has been the traditional management strategy in the area, although recent policies to promote sustainable practices within the Nature 2000 network have led to a ban on this practice. Now, straw is mainly baled and removed from the field. Incorporating the straw into the soil can be more positive, despite the increase in CH_4 emissions, because it decreases SOC losses. The EU is aiming for climate neutrality by 2050, which implies, among other things, the need to implement management practices that increase the content of recalcitrant carbon (humus) in the soil since it will remain in the soil for millions of years, compared to more mineralizable fractions. This will enhance the soil's function as a carbon sink. Water and straw management and the moment of straw incorporation into the soil are critical to reduce greenhouse gas impacts and increasing SOC retention capacity. Long-term field studies that experimentally measure the emissions associated with each alternative are needed to effectively promote carbon farming practices related to straw management in paddy rice, including the addition of straw biochar and future climate change projections that may affect salinity and soil carbon storage.

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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