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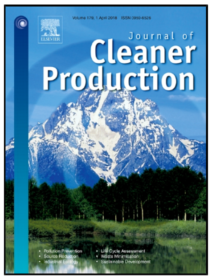

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Impact of different water and nitrogen inputs on the eco-efficiency of durum wheat cultivation in Mediterranean environments

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Abstract

The present study addresses the eco-efficiency (environmental and economic trade-offs) of durum wheat cultivation practices adopted at field level under typical Mediterranean conditions of Southern Italy. This study is based on three years of experimental data of durum wheat cultivation under three water supply regimes (full irrigation, 50% of full irrigation and rainfed) coupled with two nitrogen (N) fertilizer levels (high N, HN: 120 kg/ha, and low N, LN: not fertilized). The environmental impact assessment was based on a novel life cycle impact assessment method which quantifies seventeen midpoints (problems-oriented) and three endpoints (damage-oriented) indicators using ReCiPe 2016 model. The economic performance was evaluated using the total value added to the system's final products due to water and N use and applied management practices. Eco-efficiency was assessed as a ratio of the total value added to the environmental impact categories. The water consumption impacts were estimated in addition to the typical environmental impact categories. The high input (irrigation and fertilization) intensity systems resulted in higher agricultural production, however, produced greater impacts on water consumption, global warming, and energy-related indicators. In turn, these impact categories generated the damages to human health, ecosystem quality, and resource scarcity. The analysis demonstrated that eco-efficiency cannot be always compensated by higher yield and corresponding economic total value added. The eco-efficiency assessment indicated that agronomic practices with the low use of resources (e.g., deficit irrigation with low N) tend to have higher eco-efficiency than more intensive cultivation strategies. Hence, the sustainable crop production strategies should evolve towards the adoption of precision agriculture and optimization of water and fertilization inputs (in space, timing, and quantities) that can improve yield response to resources, environmental and economic performance. In this sense, life cycle thinking and assessment considering multiple impact categories are essential to support decision-making processes towards sustainability.

***Keywords: integrated resource management; agriculture; irrigation; LCA; environmental performance; farm sustainability**.*

1. Introduction

The global demand for wheat is expected to increase by 60% by 2050 (Dixon et al., 2009). The bulk of the projected growth in crop production will be due to the intensification of cultivation, i.e. increased fertilizer and water use, and energy consumption. At the same time, the environmental pressures posed through intensified agricultural activities will likely increase. Hence, the selection of the site-specific and resource-optimized management practices and crop varieties is to increase/stabilize yields and water productivity (Todorovic, 2016).

In the Mediterranean region, the assessment of eco-efficiency of food supply chain, resource management policies, and on-farm agronomic measures is of great importance to preserve limited natural resources and assure sustainable ecosystems functioning and resilient rural development. Therefore, the intensification of crop production has to be supported by the optimization of resource use efficiency and agronomic measures able to improve the environmental performance of the agricultural systems. In this context, the life cycle thinking is increasingly seen as a key concept for systematically analyzing farming practices, thus ensuring the transition towards more sustainable production and consumption patterns (Notarnicola et al., 2017; Sala et al., 2017).

Life Cycle Assessment (LCA) methodology is frequently used for calculation of potential environmental impacts of material and energy inputs of product or processes (ISO, 2006). Many studies, with a wide geographic context, have been conducted using the LCA approach to analyze the environmental impacts of wheat production systems. Charles et al. (2006) used a multi-impact LCA analysis of wheat crop with different intensities of production in Switzerland. Biswas et al. (2008) presented a greenhouse gas (GHG) life cycle assessment of wheat production in an Australian context. Meisterling et al. (2009) used a streamlined hybrid LCA to compare the global warming potential (GWP) and primary energy use of conventional and organic wheat production in the US. Tahmasebi et al. (2017) investigated the productivity and environmental impacts of irrigated and rainfed wheat production systems in Iran highlighting the need for better balancing between productivity and sustainability. Recently, Ali et al. (2017) have elaborated the effect of different levels of soil disturbance and nitrogen inputs on the greenhouse gas emissions of durum wheat cultivation in Southern Italy. However, these studies focused on some specific indicators of environmental aspects of wheat cultivation and did not consider the impact of different water inputs and the economic aspects of eco-efficiency. Henceforth, the development of metrics for measuring environmental impacts of a product or service system along with its economic performance is needed to explore the trade-off between economic and environmental sustainability (Georgopoulou et al., 2016).

In the recent years, the concept of eco-efficiency has been promoted to embrace ecological and economic aspects of production towards the site-specific and resource optimized management practices (Keating et al., 2010; Park et al., 2010; Todorovic et al., 2016). The eco-efficiency can be applied as a composite indicator for the evaluation of agricultural systems on the road toward sustainable intensification of production (Gadanakis, 2014).

The Mediterranean environments are characterized by chronic water shortage and irregular precipitation pattern. Thus, the adoption of supplementary irrigation and its interaction with nitrogen input are of primary importance to stabilize cereal production in the region (Oweis et al., 1998; Abi Saab et al., 2015). The present study applied a systemic analysis to assess the eco-efficiency of durum wheat cultivation under various management strategies adopted at field-level in Southern Italy. The study explored the effects of different irrigation and nitrogen inputs, and corresponding agricultural practices with special focus on environmental sustainability, and it quantified the eco-efficiency performance of the implemented management options.

ReCiPe 2016 (Huijbregts et al., 2017) the most recent harmonized life cycle impact model including seventeen midpoint and three endpoint impact categories was employed to highlight the importance of all of the potential environmental impacts. The broadest set of impact categories includes also water consumption, one of the emerging categories with the greatest interest to the LCA of agricultural production systems in the Mediterranean region. The environmental impact of water consumption is determined both at the midpoint (water consumption) and at endpoint level (damage to the ecosystems and human health). The adoption of such approach supports stakeholders and policymakers in analyzing the agricultural systems and identifying the best mitigation/adaptation options of mutual interests and for more eco-efficient agricultural production (Levidow et al., 2014; Mehmeti et al., 2016).

2. Materials and Methods

The methodology was based on a combination of the LCA (ISO, 2006) and the assessment of the Total Value Added (TVA) to the system's final products due to water and nitrogen use and applied management practices.

2.1 Goal definition, functional unit, and system boundaries

In this study, the agricultural production system corresponded to the agronomic practices adopted for durum wheat cultivation in Puglia region (Southern Italy). The production system under study considered both foreground and background systems (Figure 1), and included a set of life cycle production stages (S), namely, land preparation, i.e. soil tillage and land leveling (S1), sowing (S2), growing (S3) and harvesting (S4).

Figure 1. Insert here

The operational data (i.e. use of resources, agronomic practices, and corresponding yield response) for default operations were collected from three years of field research experiments carried out in 2005-2006, 2006-2007 and 2007-2008 at the Mediterranean Agronomic Institute of Bari (Italy). The study site and experimental setup were described in details by Albrizio et al. (2010). The scope of the present study was defined as "the cradle to the gate of the field" and included the adopted agronomic practices, i.e. different water and nitrogen inputs and corresponding energy consumption during the farming season as well as the use of pesticides and materials for equipment (tractor and irrigation system) production. Two functional units (FU) were defined as: (a) 1 ha of cultivated land, i.e. based on the land occupation, and (b) 1 t of wheat obtained under different management strategies and delivered to the farm gate, i.e. based on the product unit. All the resources, emissions and LCA (i.e. values of the selected environmental indicators) were linked to both FUs. The intended audience analysis included farmers, agricultural advisors/policy makers, water users' associations, farmers' cooperatives and environmentalists. Since co-products are not harvested in this study, no allocation criteria were used.

2.2 Life Cycle Inventory (LCI) flows modeling

In this study, wheat cultivation implied crop inputs (i.e. seeds, fertilizers, water, fossil fuels, and pesticides) and corresponding grain yield achieved (Table 1). These data were used in the life cycle impact assessment (LCIA) stage to understand and evaluate the magnitude and significance of the potential environmental impacts of the adopted management practices. Wheat eco-efficiency performance was assessed under six management strategies (Table 1), i.e. three water supply regimes (100%W, 50%W, R corresponding to full irrigation, 50% of full irrigation and rainfed) coupled with two N fertilizer levels (high N, HN: 120 kg/ha and low N, LN: not fertilized and relying only on N available in the soil at sowing time).

Table 1. Insert here

The N fertilizer was applied as ammonium sulfate (21% of N) at the beginning of the tillering phase and as ammonium nitrate (26–27% of N) at the beginning of stem elongation (Albrizio et al., 2010). The full irrigation and 50% irrigation treatments received, respectively, 1660 and 830 m^3/ha of water. The energy needed for irrigation was calculated using a total dynamic head of 8 bar (average pressure required for water withdrawal, delivery, and on-field supply), pump efficiency 0.7 and motor efficiency 0.35. Pesticides were applied regularly for all treatments with a quantity of 1 kg/ha, which corresponds to the common agronomic practices in Southern Italy.

The emissions generated from crop production were specified as those from the foreground system - specifically occurring inside the system, i.e. field activities, and environmental emissions - and those from the background system, which included all other activities that deliver resources, energy, and materials to the foreground system (Figure 1).

2.2.1 Foreground system analysis

The foreground emissions are related to agricultural activities which embraced (1) land preparation, (2) sowing, (3) growing (water, fertilizer and pesticide inputs), and (4) harvesting. Moreover, the foreground emissions included the use of fuel and other energy sources for the operation of agricultural machinery (irrigation pumps and tractors). The emissions of air pollutants due to fuel combustion in agricultural machinery were estimated as a product of the mass of fuel consumed in each process, and pollutant emitted by the combustion of 1 kg of diesel fuel (most air emissions). Environmental impacts of agricultural operations are deeply affected by a set of parameters such as soil texture, field shape ratio, declivity, climatic issues such as temperature and rainfall (Lovarelli et al., 2017). However, local characteristics for each farm are difficult to obtain, thus, the Ecoinvent emission factors (Corrado et al., 2017) were used for modeling agricultural operations since was considered more useful in addressing geographic representativeness (world average). The amount of machinery (module tractor production) needed for a specific process was derived from Nemecek and Kagi (2007) using tractor mass of 3600 kg and lifetime of 7000 hours. The fertilizer field emissions were estimated from the amount of fertilizer applied and the corresponding pollutant emitted. The most important releases from nitrogen (N) in crop production, such as nitrous oxides (N_2O), ammonia (NH_3) and nitrates (NO_3), were modeled. Two models proposed by Brentrup et al. (2000) and IPCC (2006) are mostly used by LCA practitioners to calculate emissions from fertilizers in agricultural LCA studies (Schmidt Rivera et al., 2017). Exploration of the pertinence of models used to estimate field emission was recommended by Corrado et al. (2017), however, the most appropriate way to model agricultural practices remains still under debate. The emission of N_2O from the application of the N-fertilizer to the soil was calculated as direct and indirect contributions according to the Intergovernmental Panel on Climate Change (IPCC, 2006) guidelines and methodology described by (Koeble, 2014). The N_2O loss rate was calculated as follows:

$$N_2O - N = (FSN + FCR) \times EF_D + FSN \times FRA_{GASF} \times EF_{ATD} + (FSN + FCR) \times FRA_L \times EF_L \tag{1}$$

where: $\mathbf{N_2O}$ is annual N_2O emission produced within each management strategy [kg N_2O/y], $\mathbf{F_{SN}}$ is the amount of synthetic fertilizer applied [kg N/ha]; $\mathbf{F_{CR}}$ is the amount of N crop residues above-ground and below-ground [kg N/ha]; $\mathbf{EF_D}$ is the emission factor for N_2O emissions from N inputs [kg N_2O–N/kg N]; $\mathbf{Frac_{GASF}}$ is the fraction of synthetic fertilizer N that volatilizes as NH_3 and NO_x [kg N volatilized/kg N]; $\mathbf{EF_{ATD}}$ is the emission factor for N_2O emissions from atmospheric deposition of N on soils and water surfaces [kg $N–N_2O$/kg $NH_3–N + NO_x–N$ volatilized]; $\mathbf{Frac_L}$ is the fraction of all N added to/mineralized in managed soils in regions where leaching/runoff occurs and which is lost through leaching and runoff [kg N/kg of N additions]; $\mathbf{EF_L}$ is the emission factor for N_2O emissions from N leaching and runoff [kg $N_2O–N$/kg N leached and runoff].

The IPCC (2006) uses default values to account for these emissions, independently of the type of fertilizer, environment, crops, and management practices. To account for site-specific conditions (environmental, soil characteristics, agricultural management practices, and fertilizer type), direct N_2O and NO emissions were calculated based on the regression model (Eq. 2) developed by Stehfest and Bouwman (2006). The indirect pathways (leaching/runoff and volatilization) calculations were based on the IPCC TIER1 method for all nitrogen sources. Site-specific measures of emissions from agricultural systems can increase the representativeness of the results (Corrado et al., 2017). Direct emissions of N_2O or NO, expressed in kg/ha of N over the time period covered by the measurements, were estimated as:

$$\log\left(N_{emission}\right) = \mathrm{A} + \sum_{\mathrm{i}=1}^{\mathrm{n}} \mathrm{E_i} \tag{2}$$

where A is a constant; E_i is the effect value for factor i; and n is the total number of factors (e.g. soil organic carbon content, soil pH, soil texture, vegetation type, etc.). Environmental parameters adopted for the calculations of N_2O emission were fertilization rate (Table 1), soil organic carbon content (1-3%), soil pH (5.5-8), texture (medium), climate (subtropical dry climate), vegetation class (cereals). The amount of nitrogen in crop residues (Eq. 3) was calculated from crop area and yield data (Koeble, 2014):

$$\mathrm{F_{CR}} = \left(1 - \mathrm{F_B} \times \mathrm{C_f}\right) \times \mathrm{AG_{DM}} \times \mathrm{N_{AG}} \times \left(1 - \mathrm{F_R}\right) + (\mathrm{AG_{DM}} + \mathrm{Yield} \times \mathrm{DRY}) \times \mathrm{RBG_{BIO}} \times \mathrm{N_{BG}} \tag{3}$$

where: Yield is the annual fresh yield of the crop [kg/ha]; DRY is the dry matter (d.m.) fraction of harvested product (kg d.m./kg fresh mass]; F_B is the fraction of crop area burnt annually [ha/ha]; AG_{DM} is the above-ground residue dry matter AGDM [kg d.m./ha]; F_R is the fraction of above-ground residues removed from field (kg d.m./kg AGDM); C_f is the combustion factor [dimensionless]; RAG is the ratio of above-ground residues dry matter to harvested dry matter yield for the crop [kg d.m./kg d.m.]; NAG is the N content of above-ground residues [kg N/kg d.m.].

The fraction of crop residues removed from the field and on-field burning were assumed to be zero. Nitrate leaching (Eq. 4) was estimated using the SQCB-NO_3 model (Nemecek and Schnetzer, 2012) considering a soil clay content of 30%, soil root depth of 0.7 m and adjusting the nitrogen uptake according to different management scenarios and yields.

$$N = 21.37 + \frac{P}{c \times L} \times [0.0037 \times FSN + 0.000061 \times Norg - 0.00362 \times U] \quad (4)$$

In Eq.4, N is leached NO_3-N [kg N/(ha·y)]; P is precipitation + irrigation [mm/y]; c is the clay content [%]; L is the rooting depth [m]; FSN is the nitrogen supply through fertilizers [kg N/ha]; Norg is nitrogen in organic matter [kg N/ha]; U is nitrogen uptake by crop [kg N/ha].

For seeds and pesticides, the emissions from product manufacturing and transportation to farm gate (15 km) were included in the analysis (i.e., they were treated as background impacts). The pesticide emissions were considered by applying default Ecoinvent inventories (Pesticides, unspecified at regional storage for background and Application mix, pesticides for foreground). This practice is likely to overestimate the environmental footprint of these processes, however, was considered an important assumption due to the lack of site-specific data and lack of understanding of cause-effects of such processes.

2.2.2 Background system analysis

The relative contribution of the background system was evaluated based on the respective resource input of infrastructure, seeds, fuel, electricity and generated background data-sets taken from Ecoinvent LCA database (v.3). Data for production and assembly of irrigation infrastructure were retrieved from the Australian Life Cycle Inventory Database Initiative (ALCAS, 2017). Ten years were taken as the lifetime of the irrigation system (assumed to support the hose–move sprinklers with 4 bar operating pressure). A 25 kg mass of irrigation pump was assumed (ALCAS, 2017). Typical distances for the main input materials (fertilizer, pesticides, and seeds) were calculated from typical distances in Nemecek and Kagi (2007).

Table 2. Insert here

2.3 Life Cycle Impact Assessment (LCIA)

The most recent harmonized life cycle impact model ReCiPe 2016 (Huijbregts et al., 2017) was applied to estimate the environmental impacts at midpoint and endpoint level. The midpoint approach is often the most preferred for evaluating environmental systems since has a stronger relation to environmental flows, low uncertainty, and is more familiar among researchers (Huijbregts et al., 2017; Yi et al., 2014). On the other hand, endpoint approach provides information on the environmental relevance of environmental flows (Bare et al., 2000). The midpoints indicate the contribution of a product to a specific environmental impact and are considered to be linked in the cause-effect chain (environmental mechanism) of an impact category (e.g. climate change and acidification). Endpoints are defined as the final damage to the human health, ecosystem quality and resource availability, which are caused by the various environmental effects at midpoint level. Figure 2 provides a schematic representation of the way in which an environmental damage calculation can be conducted using ReCiPe 2016 model framework.

Figure 2. Insert here

The following mid-point environmental impact categories were considered: Global warming potential (GWP), Stratospheric ozone depletion (ODP), Ionizing radiation potential (IRP), Photochemical oxidant formation - human health (HOFP), Photochemical oxidant formation - ecosystem quality (EOFP), Human toxicity potential - cancer (HTP_c), Human toxicity potential - non-cancer (HTP_{nc}), Terrestrial eco-toxicity potential (TETP), Freshwater eco-toxicity potential (FETP), Marine eco-toxicity potential (METP), Freshwater eutrophication potential (FEP), Fine particulate matter formation (PMPF), Terrestrial acidification (TAP), Agricultural land occupation potential (LOP), Water consumption potential (WCP), Mineral resource scarcity (SOP), and Fossil resource scarcity (FFP). Damage to human health (HH), ecosystem quality (ED) and resource availability (RA) were quantified on the endpoint level. The ReCiPe 2016 hierarchist perspective (without any weighting) was used since it is based on the most common policy principles with regard to time-frame (100-year timeframe is the most frequently used) and referenced to in the ISO standards on LCA (Goedkoop et al., 2013). The SimaPro software (v.8) was applied to assist in building a representative model of the physical system and to evaluate the environmental performance of the selected management practices.

2.4 Total Value Added due to water use and adopted management practices

The economic performance was measured using the TVA due to water and fertilizer use and adopted management practices (Todorovic et al., 2016; Mehmeti et al., 2016). Table 3 shows the main economic parameters adopted in this study to assess the TVA.

Table 3. Insert here

3 Results and discussion

The results are presented in terms of: (i) environmental impact of wheat cultivation, (ii) detailed analysis of the contribution of different cultivation activities to each environmental impact category, (iii) economic performance of wheat production, and (iv) eco-efficiency indicators.

3.1 Environmental performance of wheat production

The environmental impact of wheat cultivation for each of the six management strategies – scenarios are presented for twenty environmental impact indicators (seventeen midpoints and three endpoints). The results are expressed in two functional units, i.e. per area cultivated (Table 4) and per mass of product (Table 5). The former unit represents the system intensity, while the latter depicted the efficiency of the wheat production system (Charles et al., 2006). The use of a multi-index to express the results for the same category facilitated the comparison across management strategies and provided clearer support for decision-making.

For the area-based indicators (Table 4), the greatest emissions were obtained in the highest input-intensity system (full irrigation and HN input), whereas the lowest emissions were reported under the rainfed cultivation with low N input, i.e. in the lowest input-intensity system. The global warming (GWP), stratospheric ozone depletion (ODP) and terrestrial acidification potential (TAP) impacts were the most relevant indicators and they increased when moving from rainfed with low nitrogen (R+LN) to rainfed with high nitrogen (R+HN) system. In the irrigated strategies with low input of nitrogen (i.e. 50%W+LN and 100%W+LN), the water consumption (WCP), fossil resource scarcity (FFP) and human carcinogenic toxicity (HTPc) are the most relevant impact categories.

Table 4. Insert here

Table 5. Insert here

For the product-based indicators (Table 5), the environmental performance varied differently across the adopted management systems and it is based on energy input and yield achieved. The results showed that rainfed wheat presents lower impacts than irrigated for almost all categories (except for FETP and TETP versus 100%W strategy). However, product-based indicators indicated better performance of the system when N was applied with irrigation, i.e. when the positive impact of N on yield was greater than in the case of rainfed cultivation (R+HN). Correspondingly, the high input-intensity system (100%W+HN) had low-energy input requirement and corresponding environmental footprint due to higher tons of product harvested compared with R+HN and 50%W+N.

In recent years, several studies of LCA have been carried out on the environmental assessment of winter wheat production. Charles et al. (2006) estimated the environmental impacts of different wheat cultivation treatments. Global warming potential, acidification potential, and eutrophication were estimated to be 2417 kg CO_2-eq/ha, 17.8 kg SO_2-eq/ha and 3.47 kg PO_4-eq/ha, respectively, under standard treatment with 140 kg/ha of N fertilizer input. Brentrup and Palliere (2008) estimated the carbon footprint of winter wheat to be 2,516, 1,569 and 295 kg CO_2-eq/ha for the economic optimum N rate (190 kg/ha), the 50% optimum (90 kg/ha), and without N input, respectively. In New Zealand, Barber et al. (2011) measured GWP of one ton of arable product to the farm gate by means of LCA as 340 kg CO_2-eq/t of grain and 2,820 kg CO_2-eq/ha. In the Australian context, Brock et al. (2012) estimated a total carbon footprint of 200 kg CO_2-eq/t of wheat at the farm gate, based on 3.5 t/ha of grain yield. For a higher yielding crop (5.0 t/ha), total emissions of 150 kg CO_2-eq/t were reported. For Italy, Hayer et al. (2008) estimated the GWP of wheat to be about 2768.59 CO_2-eq/ha. Under Mediterranean conditions, Ali et al. (2017) estimated the GHG of wheat ranging from 839 to 1994 kg CO_2-eq/ha, depending on the adopted tillage system. In a meta-analysis of wheat LCA studies in different countries, Achten and Van Acker (2016) showed substantial differences. Average GWP was found to be 0.56 kg CO_2-eq/kg ranging from 0.3 to 1.07 kg CO_2-eq/kg. The acidification potential and eutrophication potential were estimated to be 3.05 kg SO_2-eq (range from 1.95 to 6.35) and 1.67 kg PO_4^{3-}-eq (range from 0.34 to 3.04), respectively. Therefore, from the above studies, it can be inferred that the life-cycle estimates inevitably have large variability, depending on the inventory data and modeling approaches (Corrado et al., 2017).

3.2 Analysis of process contribution in environmental performance of wheat production

Figure 3 presents the detailed analysis of the contribution of different cultivation activities to each environmental impact category. In the case of high input systems (Figs. 3b. 3d e 3f), the foreground system generated a higher environmental footprint due to higher input necessary to increase production. The foreground system emissions were notable particularly for GWP, TAP, and FEP, especially in the nitrogen adopted management strategies. Over the entire crop production cycle, the wheat cultivation stage was the main contributor to the freshwater consumption (WCP), land use (LOP) and stratospheric ozone depletion (ODP) impact categories. The importance of each process is more extensively illustrated in the subsequent sections of this paper.

Figure 3. Insert here

3.2.1 Effect of seeding material

Seed production and transportation to farm gate were responsible for a notable share of environmental impacts in rainfed wheat-production systems inducing up to 70% of the impact as in the case of human non-carcinogenic toxicity (HTPnc) impact category (Figure 3). In the case of fertilized management strategies, the seed production impacts were overcome by the fertilizer-related impacts due to more energy-intensive background processes and on-farm emission releases. However, they remain considerably high for certain categories (e.g. FEP, HTPnc, SOP). Ghorbani et al. (2011) estimated that seed production was responsible for 23.6% of energy demand in rainfed wheat production systems and 11.08% in irrigated wheat production. Brock et al. (2012) reported that the wheat seed production contributed by 2% to total GWP impacts. In a meta-analysis of wheat LCA studies, Achten and Van Acker (2016) concluded that seeding material and sowing process steps were responsible for 4% of fossil energy demand and 3% of GWP of the production of 1 kg of wheat grain. Ali et al. (2017) analyzed the GHG emission of durum wheat in Southern Italy for a seeding rate of 200 kg/ha and estimated that seeds, on average, were responsible for 9% of total GHG impacts, with a range from 6% to 22%, depending on the adopted tillage system and fertilization regime. Tahmasebi et al. (2017) estimated a lower share of seeds contribution on total GHG impact for irrigated (1.2-1.8%) and rainfed (2.2-3.5%) cultivation. The contribution of seeds to total GHG impact has increased from the high yield, over the medium yield to the low yield group of farmers.

This study showed the relative importance of seed production (mainly in low-input systems), indicating that future research should focus on the analysis to further explore the effects of seeding rate on yield and eco-efficiency performance.

3.2.2 Effect of fertilizer regime

The application of fertilizers (and especially of nitrogen) is crucial to enhance the yield of cereals (Albrizio et al., 2010). The application of nitrogen in the dry land management system generated 33% additional agricultural production (R+HN, Table 1). However, its consumption is associated with excessive environmental problems such as acidification, eutrophication, human toxicity and ecological toxicity (Brentrup et al., 2004; Brentrup and Palliere, 2008; Nemecek and Kagi, 2007).

The results of this study indicated that management practices with the high use of N caused more detrimental impacts on the environment (Table 4). The analysis showed that the utilization of nitrogen fertilizer had the greatest impact on GWP, TAP, ODP, and human toxicity.

The impacts were from two up to fourteen times higher – depending on the impact category – when moving from rainfed with low nitrogen (R+LN) to rainfed with high nitrogen (R+HN) system. Many emissions and impacts are related to the input of fertilizers to agricultural soil, with 56% related to ammonium nitrate and 44% to ammonium sulfate. Nitrogen fertilizer application emitted reactive nitrogen substance in the environment in the form of ammonia (NH_3), nitrate (NO_3), dinitrogen monoxide (N_2O), and nitrogen oxide (NO_x) which contributed to the environmental burdens. The type of fertilizer mainly influences the magnitude of these fluxes. Ammonia volatilization was a major N loss which dominated terrestrial acidification (TAP) impacts. More intensive wheat production generated a higher contribution of NH_3 emissions to the total TAP since the NH_3 emission rates increase with increasing N fertilizer rates (Brentrup et al., 2004).

The N_2O emissions arising from cropping system are closely related to the N input (Table 6) and played an important role in the environmental footprint of fertilizer application (1 kg N_2O = 298 kg CO_2-eq; on a mass basis). Hence, greater application rates of nitrogen fertilizers, which are mainly the source of nitrous oxide (N_2O) with high GWP, generated their greater contribution to the GWP (Skowrońska and Filipek, 2014). In low nitrogen management systems, crop residues were the major contributors (~80%) of field N_2O emission. From R+LN to R+HN, the nitrous oxide emission was increased by 80% due to on-field emission, and about 48% of emission was attributed to nitrogen application (Table 6). Shifting from low (e.g. R+HN) to high input systems (50%W+HN or 100%W+HN), the crop residue N_2O emission was higher due to the higher yield obtained. In the case of GWP, N_2O from crop residues accounted for 32% in the rainfed management system (R+LN), 14% in R+HN system and about 15% in combined water and fertilizer strategies. In any strategy, N_2O field emission is the predominant source of the total GWP. Total N_2O emissions from all sources accounted for 28.3 to 52.4% of GWP, with an average of about 43.3%. Ali et al. (2017) have reported an average of 37.4%, while Yang et al (2015) indicated a range from 28.4 to 54.2% of total GWP. Tahmasebi et al. (2017) reported a share of 29.2 and 45.2% of total GWP in irrigated and rainfed production systems, respectively.

Table 6. Insert here

Nitrate (NO_3) leaching from agricultural soils represents a substantial loss affecting eutrophication of groundwater. Conceptually, greater N fertilizer rates and amount of water passing through and below the root zone would lead to increased NO_3 leaching potential, but this is not always the case. In this study, we estimated a reduction of NO_3 emissions for both LN and HN treatments when moving from rainfed to irrigated wheat cultivation. In irrigated strategies with low input of nitrogen (i.e. 50%W+LN and 100%W+LN), the NO_3 emissions were reduced by 10% and 37%, respectively, compared with low N input rainfed. Similarly, in the case of high N input, the NO_3 of 50%W and 100%W strategies was by 12 and 22%, lower with respect to rainfed, respectively (Figure 3). Soil characteristics and moisture conditions are the two main factors determining the nitrate leaching fraction (Zeinali et al., 2009). When there is no or low drained water below the root zone (such as low-rainfall years, nitrate leaching can be limited (in light- and medium textured soil) or even zero (in fine-textured soil). However, nitrate leaching depends also on the distribution of N applications during the season and their relation to the watering events (referred to both rainfall and irrigation). In fact, in a five-year observation study, Yang et al. (2015) highlighted that the annual nitrate leaching showed weak correlations with annual rainfall, annual irrigation and even the sum of them. In any case, since the fertilizer productivity was strongly related to soil moisture, the fertilizer applications should be well balanced with rainfall and irrigation events in order to increase crop productivity (Alexandratos and Bruinsma, 2012).

Similarly, with fertilizer application, environmental impacts are directly linked also with the production of mineral fertilizers in the pre-farm stage and to their transportation to a lesser extent. During fertilizer manufacturing, fossil energy use and losses of nitrogen compounds to the environment contribute to several environmental impacts, demand for abiotic resources and toxicity related indicators in particular. Carbon dioxide (CO_2) and nitrogen monoxide (NO), released from fossil fuels used in fertilizer manufacturing (e.g. ammonia production and nitric acid), have also the noticeable effect on effect on environmental footprint. In toxicity-related categories, pesticide manufacturing and transportation were important as well, mainly in low input systems.

As a whole, the use of nitrogen fertilizer was a significant process that contributed to a relevant share to almost all environmental categories analyzed in this study. This research supported the key finding of other studies that N fertilizers make considerable contributions to the overall environmental footprint of crops (Ghorbani et al., 2011; Nemecek et al., 2011; Ali et al., 2017). Fertilizer production and use are strongly interlinked, hence, the improvement of the eco-efficiency of wheat cultivation is highly recommended by optimizing the management practices and minimizing the use of external production inputs, such as purchased fertilizer and pesticides. Possible strategies to enhance wheat productivity and eco-efficiency include precision N management (timing, dose, type of fertilizer), and selection of environmentally-friendly technologies that optimally utilize resources for fertilizer production and use (Skowrońska and Filipek, 2014).

3.2.3 Effect of irrigation regime

Supplemental irrigation of winter wheat is a common practice in many Mediterranean areas because the region is characterized by a semi-arid climate and erratic precipitation pattern during the winter-spring season. The application of irrigation could be particularly beneficial before flowering in order to avoid yield losses and stabilize cereal production (Karam et al., 2009; Albrizio et al., 2010). Ventrella et al. (2012) reported that irrigation significantly increased biomass and yield of winter durum wheat grown under climatic conditions of Southern Italy. In the future, water scarcity will likely increase (Saadi et al., 2015), which may prioritize water use for irrigation of summer horticultural and industrial crops rather than winter cereals. Therefore, understanding the effects of water application on yield and environmental performance becomes an essential step in planning a sustainable irrigation strategy.

In the case of low N input, the rainfed system (R) generated 655.8 kg CO_2-eq/ha or 190.1 kg CO_2-eq/t (Table 4 and Table 5). By increasing irrigation water supply, the environmental effects due to water use and energy consumption are further increased. This cause-effect chain of increased water consumption, caused impacts on human health and natural environment (both terrestrial and freshwater quality) as demonstrated in Table 4 and Figure 4. The inclusion of the impact of freshwater use is critical for endpoint categories (mainly human health), which confirmed the importance of considering both midpoint modeling and endpoint modeling when performing LCA of agricultural systems.

Figure 4. Insert here

The irrigation water use by crop represented more than 55% of total water consumption, while the rest was shared among other different processes. The GWP increased with applied irrigation volume and they were by 52 and 61% per ha greater for 50%W and 100%W, respectively, with respect to the rainfed system. When the comparison between three water regimes was made per 1 t of yield produced, the CO_2 emissions of 50%W and 100%W were by 42 and 23% greater than rainfed cultivation, respectively. The difference in terms of functional units was due to non-linearity of crop-response-to-water function. In fact, in terms of yield, the benefits of irrigation were greater than the negative impacts due to additional water use.

The irrigated system generated supplementary environmental impacts because of the energy use linked with irrigation water withdrawal, delivery and application, and associated irrigation infrastructures. The consumption of diesel fuel for irrigation was responsible for the majority of the impacts, while the environmental impact of irrigation infrastructure was almost negligible due to the relatively long lifetime of irrigation system (assumed to be 10 years in this study). Total energy consumption and corresponding environmental performance of irrigation are related to the quantity of water applied, the groundwater depth, the irrigation method – equipment, and type of energy system (Mila-i-Canals, 2003; Mehmeti et al., 2016). Changes in field practices could reduce the environmental impacts of irrigated agriculture and, at the same time, diminish the costs of production. For example, reducing the depth of groundwater pumping by 20% will reduce environmental footprint in the same order of magnitude. Moreover, for example, the adoption of electric pumps instead of diesel pumps reduces environmental burden since it eliminates foreground and background impacts of production and use of diesel for irrigation (Mehmeti et al., 2016). However, the trade-offs must be pursued. From the analysis of the Ecoinvent data, it is found that high environmental effects for toxicity-related impact categories could be expected from the electricity generated by the Italian grid for a single unit of energy supplied. Nevertheless, it is extremely important to investigate the results in terms of the life cycle stage where the emissions occur. A higher rate of electricity life-cycle emissions does not necessarily result in a greater local environmental impact because the power production does not take place within the farm system boundaries. Development of eco-innovative irrigation infrastructures in Southern Italy (and most of the Mediterranean regions) remains an important challenge. In this context, an effective institutional and legal framework and funds for initial implementation are needed.

3.2.4 Effect of agricultural operations

The interest in environmental impacts linked to agricultural machinery has increased enormously (Lovarelli et al., 2017). The environmental impacts of the on-field operations are derived from the use of machinery (mainly due to fuel consumption), and from the production and maintenance of machines. The most important factor is the use of fossil fuel for machinery operations including plowing, harrowing and combine harvesting (Fallahpour et al., 2012; Ali et al., 2017). Even so, it is commonly argued that a complete LCA study should comprise the production of machinery and other capital equipment because they can have a relevant share of the overall impacts (Mila-i-Canals, 2003; Frischknecht et al., 2007).

In this study, under rainfed management strategy (R), the use of agricultural machinery for different agronomic practices caused, on average, 13.2% impacts in foreground system due to fuel combustion, and generated about 3.4% in background system due to fuel production. These effects were further enhanced in the high input systems since a higher diesel consumption was observed due to fertilization spraying. Fuel consumption and emissions of harmful components can be reduced only by complex optimizing of technological processes and tractor operating modes in real working conditions (Janulevičius et al., 2013).

The production of agricultural machinery (i.e. tractors) significantly influenced the contribution of the background system to nearly all impact categories in low-input agricultural systems with low fertilizer application (Figure 3). Some sensitive impact categories were water consumption (28.17%), human carcinogenic toxicity (33.1%) and mineral resource scarcity (37.37%). In the high input systems coupled with high fertilization strategies, the machinery emissions were overcome by fertilizer-related emissions due to more energy-intensive processes and field emissions. Tractor production-related environmental impacts depend on working time, lifetime and implemented weight. In general, the inclusion of infrastructure, for example, is noteworthy for the impact categories of toxicity indicators, which are associated with energy-intensive manufacturing processes.

3.3 Economy of wheat production

The quantified economic performance of wheat production under the evaluated management practices is presented in Table 7. The economic analysis showed that the costs of wheat production range between 379 and 784.1 €/ha. In the case of rainfed cultivation, the TVA was estimated to be 311 €/ha or 85.4 €/t. The application of deficit or full irrigation with low fertilization input (50%W or 100%W) generated lower TVA on the area basis, as yield increase cannot compensate for the cost for such strategies. On the other hand, the optimal combination of water (i.e. full irrigation) and fertilization regime can maintain optimum yield and increase economic benefits. In the case of irrigated and fertilized wheat cultivation (either 50%W+HN or 100%W+HN), the TVA on the area basis was increased by more than 30% due to the higher yield obtained. However, on the product basis, the greatest TVA was found under rainfed cultivation with the application of nitrogen (R+HN) due to well-balanced income and costs of production.

The results confirmed that irrigation remains a profitable activity under the current economic situation and actual wheat market price. In general, the main role of irrigation is to provide greater income stability, although wheat rainfed farming systems in most of the Mediterranean area are highly efficient with significant positive effect on benefit-cost ratio and energy efficiency (Ghorbani et al., 2011). Nevertheless, it is important to emphasize that the profitability of irrigation can change from one year to another depending on the wheat market price, precipitation quantity, and distribution during the growing cycle. In this context, the expected variability of precipitation pattern and its reduction in the future over most of the Mediterranean area will pose additional constraints to rainfed cultivation and promote supplemental irrigation for sustainable agricultural production (Tanasijević et al., 2014; Saadi et al., 2015).

Table 7. Insert here

3.4 Quantification of eco-efficiency performance

The Eco-Efficiency Indicators (EEI) defined as ratios of the economic performance (total value added, TVA) to each of environmental impact categories are expressed on the area (Table 8) and product basis (Table 9) for different water and N management strategies. On the area basis, for the majority of impact categories, the rainfed cultivation reflected more than twice higher eco-efficiency compared to other strategies. The analysis demonstrated the importance of the adopted irrigation practices and showed that a product under rainfed conditions might have a greater eco-efficiency than a product under irrigation or fertilizer regime. The latter becomes less eco-efficient because the environmental impacts linearly increase with the augmented water and N supply and corresponding agronomic practices, which cannot be always compensated by higher yield and total value added (for example in the case of 100% irrigation and fertilizer regime). The product based indicators showed better eco-efficiency performance for the rainfed system and for the system when N was applied with irrigation (100%W+HN) due to better economic performance as a consequence of higher yield of the product harvested.

Table 8. Insert here

Table 9. Insert here

Obviously, yields of irrigated crops were well above those of rainfed ones; however, the key question is whether sufficient water is available to satisfy the growing needs of agricultural users since water scarcity is becoming a real threat to the sustainability of irrigated agriculture in Southern Mediterranean countries. Hence, the expansion of irrigation activities will require careful assessment of water requirements and withdrawals, and continuous assessment of models for water resources use. It should be noted that agricultural systems are dynamic in space and time, therefore, their eco-efficiency alter with adopted agronomic practices, climatic conditions (precipitation pattern and air temperature), soil characteristics, water availability, economic parameters and other external factors (Todorović et al., 2016). Moreover, impact analysis shows that results depend to a substantial degree on the functional unit chosen and on the goal of the LCA. Henceforth, this study confirmed the recommendations of other studies (Charles et al., 2006; Nemecek et al., 2011) to use two functional units simultaneously, to better assess the efficiency and intensity of wheat production systems. While there is no "optimal" strategy for eco-efficiency of agricultural systems, this study highlighted that rainfed wheat production systems could lead to the enhancement of sustainable agriculture in dry and semi-dry climates due to higher energy efficiency and eco-efficiency ratio for the majority of impact categories analyzed.

4 Conclusions

Modern strategies for agricultural production should decouple economic benefits (biomass and yield growth) from undesirable environmental impacts, which requires a better understanding of emissions, production costs, market trends and their drivers. In this study, an in-depth quantitative analysis of combined environmental and economic impacts of wheat production under different water and nitrogen regimes was performed following a "cradle-to-gate" approach. The quantified results identified resource-optimized management practices, which can reduce emissions and corresponding environmental impacts, and, simultaneously, improve productivity.

This study underlines the inadequacy of single-criterion approaches to LCA of wheat production, where only the global warming potential impact category is considered. Other impact categories, such as water consumption, acidification potential, land occupation and toxicity related indicators, should also be evaluated to accomplish a holistic environmental analysis and to account for the interactions and synergies between midpoint and endpoint impacts.

The highest emissions were obtained in the highest input-intensity system (full irrigation and high N input), whereas the lowest was reported in the lowest input-intensity system (rainfed with low N input) with a reduction from two to fifteen times, depending on the impact category. Although gross economic return per ha in rainfed wheat production systems was less than in full input systems, the findings of this research provided evidence that higher eco-efficiency performance is obtained when agronomic practices with the low use of resources (water and nitrogen) are applied.

This study emphasized the importance of adopted management practices in Mediterranean agricultural systems and suggested that the optimization of water and fertilization inputs (in terms of space, timing, and quantities) can improve yield response to resources and increase eco-efficiency. Hence, in the future, agricultural production should be driven by detailed monitoring of soil-canopy-atmosphere continuum and adopted the precision agriculture concept to maximize productivity and profitability while reducing the environmental burden and increasing the sustainability of cultivation. In the Mediterranean environments, mostly characterized by harsh conditions and resource scarcity, embracing environmental awareness is the most advisable way to gain in eco-efficiency and increase the ecosystem well-being.

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Figure 1. System boundaries and life cycle stages (S) adopted for the eco-efficiency assessment of on-farm wheat cultivation.

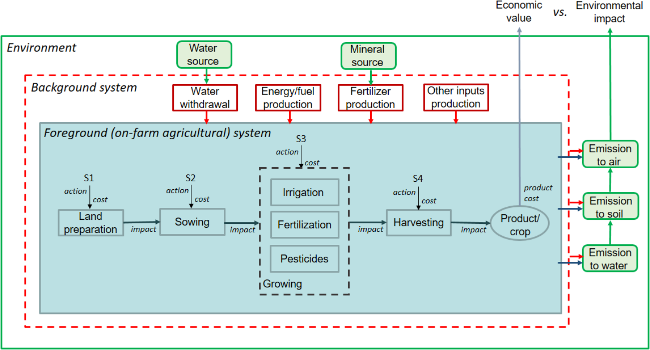
Figure 2. Schematic impact calculation in ReCiPe on both midpoint and endpoint level of wheat production systems. Environmental impact categories: Global warming potential - GWP, Stratospheric ozone depletion - ODP, Ionizing radiation potential - IRP, Photochemical oxidant formation - human health - HOFP, Fine particulate matter formation - PMPF, Photochemical oxidant formation - ecosystem quality - EOFP, Terrestrial acidification - TAP, Freshwater eutrophication potential - FEP, Terrestrial eco-toxicity potential - TETP, Freshwater eco-toxicity potential - FETP, Marine eco-toxicity potential - METP, Human toxicity potential - cancer - HTP_c, Human toxicity potential - non-cancer - HTP_{nc}, Agricultural land occupation potential - LOP, Mineral resource scarcity - SOP, Fossil resource scarcity – FFP, and Water consumption potential – WCP.

Figure 3. Detailed environmental analysis of wheat cultivation for six management strategies: (A) rainfed + low N, (B) rainfed + high N, (C) 50% of full irrigation + low N, (D) 50% of full irrigation + high N, (E) full irrigation + low N, (F) full irrigation + high N. Environmental impact categories: Global warming potential - GWP, Stratospheric ozone depletion - ODP, Ionizing radiation potential - IRP, Photochemical oxidant formation - human health - HOFP, Fine particulate matter formation - PMPF, Photochemical oxidant formation - ecosystem quality - EOFP, Terrestrial acidification - TAP, Freshwater eutrophication potential - FEP, Terrestrial eco-toxicity potential - TETP, Freshwater eco-toxicity potential - FETP, Marine eco-toxicity potential – METP, Human toxicity potential - cancer - HTP_c, Human toxicity potential - non-cancer - HTP_{nc}, Agricultural land occupation potential - LOP, Mineral resource scarcity - SOP, Fossil resource scarcity – FFP, Water consumption potential - WCP, Damage to human health - HH, ecosystem quality - ED, and Resource availability - RA.

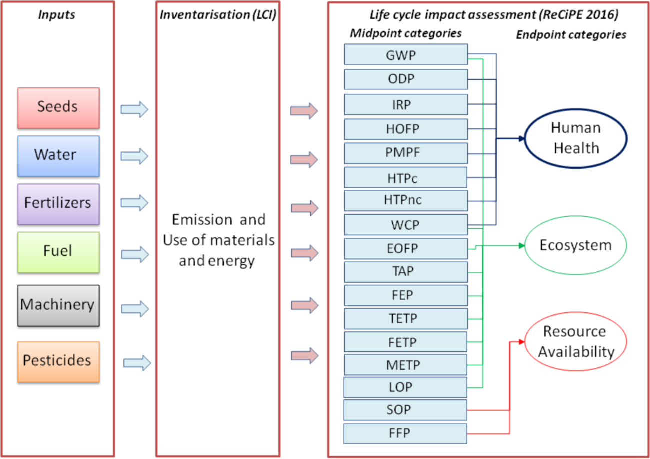
Figure 4. The contribution of midpoint categories at the endpoint level for human health (HH), ecosystem quality (ED), and resource availability (RA). Environmental impact categories: Global warming potential - GWP, Stratospheric ozone depletion - ODP, Ionizing radiation potential - IRP, Photochemical oxidant formation - human health - HOFP, Fine particulate matter formation - PMPF, Photochemical oxidant formation - ecosystem quality - EOFP, Terrestrial acidification - TAP, Freshwater eutrophication potential - FEP, Terrestrial eco-toxicity potential - TETP, Freshwater eco-toxicity potential - FETP, Marine eco-toxicity potential – METP, Human toxicity potential - cancer - HTP_c, Human toxicity potential - non-cancer - HTP_{nc}, Agricultural land occupation potential - LOP, Mineral resource scarcity - SOP, Fossil resource scarcity – FFP, and Water consumption potential – WCP.

Highlights:

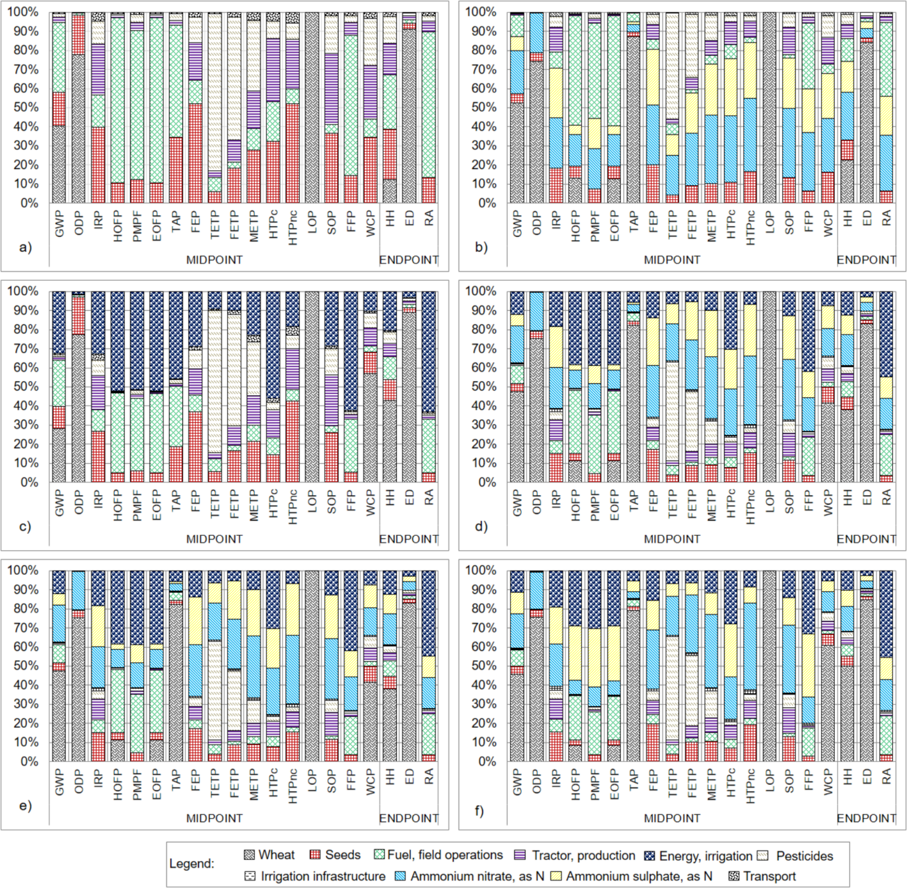
- Eco-efficiency was analyzed using twenty environmental impact categories.
- Six crop management strategies for wheat cultivation were evaluated.
- Eco-efficiency was higher under low input or optimized management practices.
- Both midpoint and endpoint impacts shall be considered to explore valuable insights.
- Agronomic life cycle based knowledge is important for sustainable farm development.



Economic value
vs.
Environmental impact
Environment
Water source
Mineral source
Background system
Water withdrawal
Energy/fuel production
Fertilizer production
Other inputs production
Foreground (on-farm agricultural) system
S1
action
cost
Land preparation
impact
S2
action
cost
Sowing
impact
S3
action
cost
Irrigation
Fertilization
Pesticides
Growing
impact
S4
action
cost
Harvesting
impact
product cost
Product/crop
Emission to air
Emission to soil
Emission to water



Inputs
Seeds
Water
Fertilizers
Fuel
Machinery
Pesticides
Inventarisation (LCI)
Emission and
Use of materials
and energy
Life cycle impact assessment (ReCiPE 2016)
Midpoint categories
Endpoint categories
GWP
ODP
IRP
HOFP
PMPF
HTPc
HTPnc
WCP
EOFP
TAP
FEP
TETP
FETP
METP
LOP
SOP
FFP
Human
Health
Ecosystem
Resource
Availability


0%
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GWP
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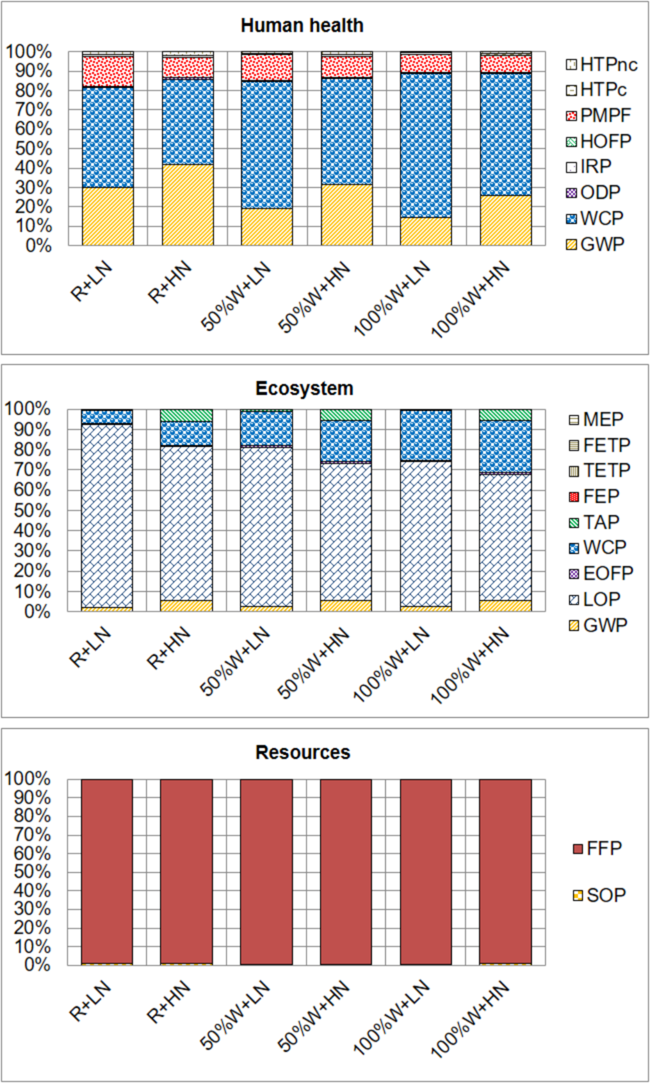
HOFP
PMPF
EOFP
TAP
FEP
TETP
FETP
METP
HTPc
HTPnc
LOP
SOP
FFP
WCP
HH
ED
RA
MIDPOINT
ENDPOINT
a)
b)
c)
d)
e)
f)
Legend:
Wheat
Seeds
Fuel, field operations
Tractor, production
Energy, irrigation
Pesticides
Irrigation infrastructure
Ammonium nitrate, as N
Ammonium sulphate, as N
Transport


Human health
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
R+LN
R+HN
50%W+LN
50%W+HN
100%W+LN
100%W+HN
HTPnc
HTPc
PMPF
HOFP
IRP
ODP
WCP
GWP
Ecosystem
100%
90%
80%
70%
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10%
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R+LN
R+HN
50%W+LN
50%W+HN
100%W+LN
100%W+HN
MEP
FETP
TETP
FEP
TAP
WCP
EOFP
LOP
GWP
Resources
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
R+LN
R+HN
50%W+LN
50%W+HN
100%W+LN
100%W+HN
FFP
SOP

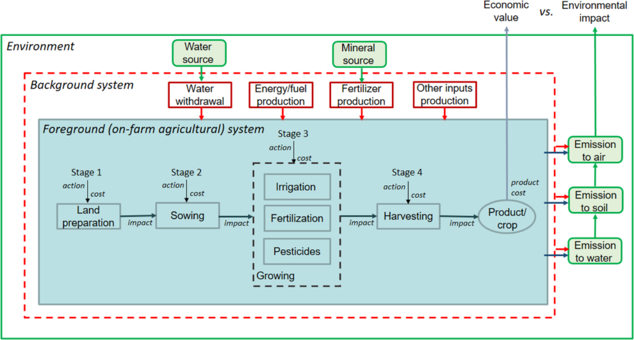
Economic value
vs.
Environmental impact
Environment
Water source
Mineral source
Background system
Water withdrawal
Energy/fuel production
Fertilizer production
Other inputs production
Foreground (on-farm agricultural) system
Stage 3
action
cost
Stage 1
action
cost
Land preparation
impact
Stage 2
action
cost
Sowing
impact
Irrigation
Fertilization
Pesticides
Growing
impact
Stage 4
action
cost
Harvesting
impact
product
cost
Product/ crop
Emission to air
Emission to soil
Emission to water

Table 1. Inventory data of wheat production under different management strategies (scenarios).

Input	Crop management strategy[a]					
	R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Rainfall (mm)	355	355	355	355	355	355
Yield (t/ha)	3.45	4.6	3.66	5.62	4.5	6.24
Seeds (kg/ha)	200	200	200	200	200	200
Irrigation						
Irrigation water (m^3/ha)	0	0	830	830	1660	1660
Diesel (L/ha)	0	0	69.17	69.17	138.33	138.33
Fertilizers and agrochemicals						
Ammonium Nitrate, as N (kg/ha)	0	60	0	60	0	60
Ammonium Sulfate, as N (kg/ha)	0	60	0	60	0	60
Pesticides, unspecified (kg/ha)	1	1	1	1	1	1
Fuel and machinery						
Diesel Fuel Operations (L/ha)	65	70	65	70	65	70
Tractors, production (kg/ha)	2.88	3.034	2.88	3.034	2.88	3.034

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N.

Table 2. Inventory data of irrigation infrastructure.

Name	Amount	Unit
Irrigation system (Hose move sprinkler irrigation)		
Drawing of pipes, steel/RER U	9.804	kg
Extrusion, plastic pipes/RER U	124.62	kg
Polyethylene, HDPE, granulate, at plant/RER U	8.9	kg
Polyethylene, LDPE, granulate, at plant/RER U	5.4	kg
Polypropylene, PP, at factory gate/RER U	0.1	kg
Polyvinylidene chloride, granulate, at plant/RER U	5.704	kg
Section bar extrusion, aluminum/RER U	1	kg
Steel, converter, low-alloyed, at plant/RER U	9.804	kg
Irrigation pump (25 kg)		
Aluminum, primary, at plant/RER U	0.5458	kg
Brass, at plant/CH U	0.0458	kg
Cast iron, at plant/RER U	18.6458	kg
Copper, at regional storage/RER U	1.0042	kg
Electricity, low voltage, at grid/RER U	194.79	kWh
Epoxy resin, liquid, at plant/RER U	0.0625	kg
Heat, at local distribution cogen 160kWe Jakobsberg, allocation energy/CH U	63.75	MJ
Heat, light fuel oil, at industrial furnace 1MW/RER U	0.0271	MJ
Heat, natural gas, at industrial furnace >100kW/RER U	6.1875	MJ
Lubricating oil, at plant/RER U	0.178	kg
Steel, converter, chromium steel 18/8, at plant/RER U	0.458	kg
Steel, low-alloyed, at plant/RER U	3.188	kg
Synthetic rubber, at plant/RER U	0.619	kg
Tap water, at user/RER U	96.250	kg
Zinc, primary, at regional storage/RER U	0.054	kg

Table 3. Cost and benefits items to assess the economic performance of the wheat production.

Category	Item	Amount	Unit
Cost	Water	0.08	€/m^3
	Irrigation system cost [a]	70	€/ha
	Seeds	0.5	€/kg
	Diesel	0.6	€/L
	Ammonium sulphate	0.23	€/kg
	Ammonium nitrate	0.25	€/kg
	Labor Cost	8	€/h
Benefits	Market price	200	€/t

[a] - Calculated with an interest rate of 5%, residual value 10% and maintenance cost of 1.5 % of total investment cost

Table 4. Area-based environmental impact indicators of wheat production for different management practices (the lowest value of each impact category is given in italic).

Impact category	Unit/ha	Crop management strategy[a]					
		R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Midpoint environmental impact categories							
Global warming	kg CO_2-eq	*655.8*	2297.6	994.2	2692.4	1052.7	2716.2
Stratospheric ozone depletion	kg CFC11-eq	*0.0126*	0.0598	0.0133	0.0627	0.0155	0.0643
Ionizing radiation	kBq Co60-eq	*23.9*	51.6	35.6	63.3	35.6	62.7
Ozone formation, Human health	kg NO_x-eq	*3.32*	5.40	6.91	9.36	6.91	9.13
Fine particulate matter formation	kg PM2.5-eq	*0.503*	0.847	1.035	1.380	1.035	1.349
Ozone formation, Terrestrial ecosystems	kg NO_x-eq	*3.36*	5.46	7.01	9.48	7.01	9.26
Terrestrial acidification	kg SO_2-eq	*2.34*	33.06	4.32	35.18	4.32	35.07
Freshwater eutrophication	kg P-eq	*0.050*	0.129	0.071	0.150	0.071	0.149
Terrestrial ecotoxicity	kg 1,4DCB-eq	*0.876*	1.298	0.966	1.388	0.966	1.382
Freshwater ecotoxicity	kg 1,4DCB-eq	*8.54*	16.75	9.48	17.69	9.48	17.62
Marine ecotoxicity	kg 1,4DCB-eq	*7.30*	20.08	9.48	22.26	9.48	22.12
Human carcinogenic toxicity	kg 1,4DCB-eq	*5.62*	16.51	12.76	23.64	12.76	23.45
Human non-carcinogenic toxicity	kg 1,4DCB-eq	*4562.8*	14414.2	5591.7	15442.6	5591.4	15351.8
Land use	$m^2 \times y$ crop-eq	*10003.0*	10007.0	10003.9	10007.8	10003.9	10007.8
Mineral resource scarcity	kg Cu-eq	*1.633*	4.482	2.285	5.134	2.285	5.095
Fossil resource scarcity	kg oil-eq	*103.8*	238.1	276.8	411.0	276.8	404.8
Water consumption	m^3	*472.9*	1005.1	1438.1	1970.2	2238.0	2759.6
Endpoint environmental impact categories							
Human Health	DALY	$\mathit{5.90 \times 10^{-7}}$	5.09×10^{-3}	4.86×10^{-3}	7.97×10^{-3}	6.69×10^{-3}	9.72×10^{-3}
Ecosystem Quality	Species × y	$\mathit{2.84 \times 10^{-8}}$	1.17×10^{-4}	1.13×10^{-4}	1.32×10^{-4}	1.24×10^{-4}	1.43×10^{-4}
Resource Scarcity	USD2013	*0.013*	94.5	121.4	171.1	121.4	168.4

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N

Table 5. Product-based environmental impact indicators of wheat production for different management practices (the lowest value of each impact category is given in italic).

Impact category	Unit/t	Crop management strategy[a] R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Midpoint environmental impact categories							
Global warming	kg CO_2-eq	*190.1*	499.5	271.6	479.1	233.9	435.3
Stratospheric ozone depletion	kg CFC11-eq	0.0036	0.0130	0.0036	0.0112	*0.0034*	0.0103
Ionizing radiation	kBq Co60-eq	*6.92*	11.22	9.72	11.26	7.905	10.04
Ozone formation, Human health	kg NO_x-eq	*0.96*	1.17	1.89	1.66	1.535	1.46
Fine particulate matter formation	kg PM2.5-eq	*0.15*	0.18	0.28	0.25	0.230	0.22
Ozone formation, Terrestrial ecosystems	kg NO_x-eq	*0.97*	1.19	1.92	1.69	1.558	1.48
Terrestrial acidification	kg SO_2-eq	*0.68*	7.19	1.18	6.26	0.96	5.62
Freshwater eutrophication	kg P-eq	*0.015*	0.028	0.019	0.027	0.016	0.024
Terrestrial ecotoxicity	kg 1,4DCB-eq	0.25	0.28	0.2641	0.25	*0.215*	0.22
Freshwater ecotoxicity	kg 1,4DCB-eq	2.47	3.64	2.59	3.15	*2.106*	2.82
Marine ecotoxicity	kg 1,4DCB-eq	2.11	4.37	2.59	3.96	*2.106*	3.55
Human carcinogenic toxicity	kg 1,4DCB-eq	*1.63*	3.59	3.49	4.21	2.836	3.76
Human non-carcinogenic toxicity	kg 1,4DCB-eq	1322.5	3133.5	1527.8	2747.8	*1242.5*	2460.2
Land use	$m^2 \times y$ crop-eq	2899.4	2175.4	2733.3	1780.8	2223.1	*1603.8*
Mineral resource scarcity	kg Cu-eq	*0.47*	0.97	0.6243	0.91	0.508	0.82
Fossil resource scarcity	kg oil-eq	*30.10*	51.76	75.62	73.13	61.500	64.87
Water consumption	m^3	137.07	218.50	392.92	350.6	497.3	442.25
Endpoint environmental impact categories							
Human Health	DALY	*1.71×10^{-7}*	1.11×10^{-3}	1.33×10^{-3}	1.42×10^{-3}	1.49×10^{-3}	1.56×10^{-3}
Ecosystem Quality	Species × y	*8.24×10^{-9}*	2.54×10^{-5}	3.09×10^{-5}	2.35×10^{-5}	2.75×10^{-5}	2.28×10^{-5}
Resource Scarcity	USD2013	*0.0038*	20.54	33.16	30.45	26.97	26.98

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N

Table 6. Dinitrogen monoxide (N_2O), ammonia (NH_3) and nitrates (NO_3) emissions breakdown for each management practice (kg/ha).

Elementary flow	Crop management strategy[a]					
	R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Direct, Synthetic fertilizer – N_2O (dir,F_{SN})	0.000	2.271	0.000	2.271	0.000	2.271
Direct, Crop residues – N_2O (dir,CR)	0.726	0.946	0.766	1.141	0.927	1.259
Atmospheric deposition – N_2O (ATD,F_{SN})	0.000	0.189	0.000	0.189	0.000	0.189
Leaching and runoff – N_2O (L,F_{SN})	0.000	0.424	0.000	0.424	0.000	0.424
Leaching, crop residues – N_2O (L,CR)	0.163	0.213	0.172	0.257	0.208	0.283
Nitrate leaching - NO_3	71.749	106.615	64.848	93.828	45.685	83.527
Ammonia volatilization – NH_3	0.000	14.568	0.000	14.568	0.000	14.568

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N.

Table 7. Economic performance of wheat production management strategies.

Item	Crop management strategy[a]					
	R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Water Cost (€/ha)	0.0	0.0	66.4	66.4	132.8	132.8
Seed Cost (€/ha)	100.0	100.0	100.0	100.0	100.0	100.0
Irrigation Cost (€/ha)	0.0	0.0	106.5	106.5	148.0	148.0
Field Operations Cost (€/ha)	39.0	42.0	39.0	42.0	39.0	42.0
Fertilizer (NPK) Cost (€/ha)	0.0	121.3	0.0	121.3	0.0	121.3
Labor Cost (€/ha)	240.0	240.0	240.0	240.0	240.0	240.0
Market Crop value (€/ha)	690.0	920.0	732.0	1124.0	900.0	1248.0
Total Cost (€/ha)	379.0	503.3	551.9	676.2	659.8	784.1
TVA (€/ha)	311.0	416.7	180.1	447.8	240.2	463.9
TVA (€/t)	85.4	101.1	45.5	72.3	43.0	62.9

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N

Table 8. Quantified area-based eco-efficiency indicators of wheat management practices (higher number means higher eco-efficiency, the highest value of each impact category is given in italic).

Impact category	€/Unit	Crop management strategy[a]					
		R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Midpoint environmental impact categories							
Global warming	kg CO_2-eq	*0.4742*	0.1814	0.1812	0.1664	0.2282	0.1614
Stratospheric ozone depletion	kg CFC11-eq	*24720.5*	6962.5	13528.4	7145.9	15521.6	7204.1
Ionizing radiation	kBq Co60-eq	*13.0*	8.1	5.1	7.1	6.8	7.6
Ozone formation, Human health	kg NO_x-eq	*93.8*	77.2	26.1	47.9	34.8	37.2
Fine particulate matter formation	kg PM2.5-eq	*618.4*	491.8	173.9	324.7	232.0	265.5
Ozone formation, Terrestrial ecosystems	kg NO_x-eq	*92.6*	76.4	25.7	47.3	34.3	36.7
Terrestrial acidification	kg SO_2-eq	*133.1*	12.6	41.7	12.7	55.6	12.7
Freshwater eutrophication	kg P-eq	*6216.3*	3224.5	2554.4	2993.4	3407.0	3527.5
Terrestrial ecotoxicity	kg 1,4DCB-eq	*354.9*	321.0	186.4	322.7	248.5	349.4
Freshwater ecotoxicity	kg 1,4DCB-eq	*36.4*	24.9	19.0	25.3	25.3	30.9
Marine ecotoxicity	kg 1,4DCB-eq	*42.6*	20.8	19.0	20.1	25.3	24.6
Human carcinogenic toxicity	kg 1,4DCB-eq	*55.3*	25.3	14.1	19.0	18.8	18.1
Human non-carcinogenic toxicity	kg 1,4DCB-eq	*0.068*	0.029	0.032	0.029	0.043	0.038
Land use	$m^2 \times y$ crop-eq	*0.031*	0.042	0.018	0.045	0.024	0.046
Mineral resource scarcity	kg Cu-eq	*190.4*	93.0	78.8	87.3	105.1	101.7
Fossil resource scarcity	kg oil-eq	*3.0*	1.8	0.7	1.1	0.9	0.9
Water consumption	m^3	*0.658*	0.415	0.125	0.227	0.107	0.174
Endpoint environmental impact categories							
Human Health	DALY	*5.28×10^8*	8.19×10^4	3.71×10^4	5.62×10^4	3.59×10^4	4.77×10^4
Ecosystem Quality	Species $\times$ y	*1.09×10^{10}*	3.57×10^6	1.59×10^6	3.40×10^6	1.94×10^6	3.25×10^6
Resource Scarcity	USD2013	*23992*	4.41	1.48	2.62	1.98	2.76

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N

Table 9. Quantified product-based eco-efficiency indicators of wheat management practices (higher number means higher eco-efficiency, the highest value of each impact category is given in italic).

		Crop management strategy[a]					
Impact category	€/Unit	R+LN	R+HN	50%W+LN	50%W+HN	100%W+LN	100%W+HN
Midpoint environmental impact categories							
Global warming	kg CO_2-eq	*0.449*	0.202	0.168	0.151	0.184	0.145
Stratospheric ozone depletion	kg CFC11-eq	*23419.3*	7770.5	12509.1	6484.0	12503.8	6108.9
Ionizing radiation	kBq Co60-eq	*12.3*	9.0	4.7	6.4	5.4	6.3
Ozone formation, Human health	kg NO_x-eq	*88.9*	86.2	24.1	43.4	28.0	43.0
Fine particulate matter formation	kg PM2.5-eq	*585.8*	548.8	160.8	294.5	186.9	291.1
Ozone formation, Terrestrial ecosystems	kg NO_x-eq	*87.8*	85.2	23.7	42.8	27.6	42.4
Terrestrial acidification	kg SO_2-eq	*126.0*	14.1	38.6	11.6	44.8	11.2
Freshwater eutrophication	kg P-eq	*5889.1*	3598.3	2361.9	2714.1	2744.6	2639.3
Terrestrial ecotoxicity	kg 1,4DCB-eq	336.2	*358.2*	172.3	292.7	200.2	284.1
Freshwater ecotoxicity	kg 1,4DCB-eq	*34.5*	27.8	17.6	23.0	20.4	22.3
Marine ecotoxicity	kg 1,4DCB-eq	*40.4*	23.2	17.6	18.3	20.4	17.7
Human carcinogenic toxicity	kg 1,4DCB-eq	*52.4*	28.2	13.0	17.2	15.2	16.7
Human non-carcinogenic toxicity	kg 1,4DCB-eq	*0.065*	0.0323	0.0298	0.0263	0.0346	0.0256
Land use	$m^2 \times y$ crop-eq	0.029	*0.046*	0.017	0.041	0.019	0.039
Mineral resource scarcity	kg Cu-eq	*180.38*	103.76	72.88	79.15	84.69	77.04
Fossil resource scarcity	kg oil-eq	*2.84*	1.95	0.60	0.99	0.70	0.97
Water consumption	m^3	*0.623*	0.463	0.116	0.206	0.086	0.142
Endpoint environmental impact categories							
Human Health	DALY	*5.00×10^{8}*	9.14×10^{4}	3.43×10^{4}	5.10×10^{4}	2.89×10^{4}	4.04×10^{4}
Ecosystem Quality	Species × y	*1.04×10^{10}*	3.98×10^{6}	1.47×10^{6}	3.08×10^{6}	1.56×10^{6}	2.75×10^{6}
Resource scarcity	USD2013	*22729.14*	4.92	1.37	2.37	1.59	2.33

[a] R+LN = rainfed + low N, R+HN = rainfed + high N, 50%W+LN = 50% of full irrigation + low N, 50%W+HN = 50% of full irrigation + high N, 100%W+LN = full irrigation + low N, 100%W+HN = full irrigation + high N