Sinisterra-Solis, NK.; Sanjuán Pellicer, MN.; Estruch-Guitart, V.; Clemente Polo, G. (2020). Assessing the environmental impact of Spanish vineyards in Utiel-Requena PDO: The influence of farm management and on-field emission modelling. Journal of Environmental Management. 262:1-12. https://doi.org/https://doi.org/10.1016/j.jenvman.2020.110325

10.1016/j.jenvman.2020.110325

Title: Assessing the environmental impact of Spanish vineyards in Utiel-Requena PDO: the influence of farm management and on-field emission modelling.

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Abstract

Environmental studies into wine from different protected designations of origin (PDO) highlight farming and packaging stages as those contributing the most to the total environmental impacts of this product. However, farming impact, not only depends on the agricultural practices but also on data quality and modelling complexity. By using the life cycle assessment methodology, a twofold goal is aimed. Firstly, to analyse the environmental profile of the most widespread viticultural practices in the Utiel-Requena PDO (Spain). The second aim is to evaluate the differences between the environmental impacts estimated by means of modelling approaches using generic information (Baseline modelling) versus those using site-specific information (Alternative modelling). As regards the agricultural practices and grape cultivars, eight systems were defined and assessed per kg of grape at the farm gate. The differences between farming systems and modelling approaches were statistically assessed. The results show that, regardless of the grape cultivar, organic systems are more environmentally friendly than the conventional ones (on average, the greatest differences occur in the ionizing radiation, marine eutrophication and land use, being the values for organic vineyards 1678%, 648% and 171% lower than those of the conventional ones, respectively), the results for the Bobal cultivar being better than those for the Tempranillo because of the higher yield (differences in yield around 1.500 kg ha^{-1}). The use of site-specific modelling approaches guarantees the precision of the analysis; however, for some impact categories, namely climate change, fine particulate matter formation, marine eutrophication and terrestrial acidification, the possibility of using general methodologies is open; in this way, the modelling efforts can be minimised, and the results would be consistent with those of more specific methodologies. The results also underline the need for a consensus within LCA practitioners on which methodologies to use in order to estimate on-field emissions taking into account both complexity reduction and accuracy improvement.

Keywords: conventional farming, organic farming, fertiliser emission, pesticide fate, environmental impacts, vineyard.

1. Introduction

Agriculture as an anthropogenic activity generates significant externalities, both positive and negative, towards the well-being of the planet (Bruinsma, 2017). Among other aspects, negative externalities are associated with significant contributions to climate change (FAO, 2014), land degradation and soil erosion (Pereyra et al., 2020; Rodrigo-Comino, Brevik & Cerdà, 2018; Prosdocimi, Cerdà & Tarolli, 2016), freshwater depletion (Villanueva-Rey et al., 2018) and pollution from plant nutrients and pesticides (Renouf et al., 2018). Within agri-food sectors, wine stands out as one of the most important in the global food market (Bonamente et al., 2016; Bosco et al., 2011), especially in the European Mediterranean countries (Spain, Italy and France), which are the main wine producers in the world.

According to data from the International Organization of Vine and Wine (OIV, 2017), in 2016, Spain was the third biggest wine producer in the world, with the largest vineyard area, and the first world exporter of wine in terms of volume. In fact, grapes are the second most important Spanish commodity, after olives (Meneses, Torres & Castells, 2016). The Utiel-Requena protected designation of origin (PDO) is an important wine supplier in Spain (CAMACCDR, 2019a). Utiel-Requena is the PDO with the greatest grape area in the region and the fifth largest in Spain, with 6% of the total grape crop (MAPAMA, 2018a). According to the *Consejo Regulador de Utiel-Requena* DO (2019), Bobal and Tempranillo are the main grape cultivars in the PDO, with 75% and 12% of the cultivated area, respectively.

Nowadays, international and governmental organizations are promoting environmental awareness in all human activities, making information available to the population and encouraging the inclusion of environmental parameters in consumer purchasing decisions (Martins et al., 2018; Schmidt Rivera et al., 2017). Along these lines, shared efforts between the different economic stakeholders have been developed. These efforts seek to improve the environmental profile of products and services from the technological point of view, creating innovative technologies which are more environmentally friendly, together with the development of methodologies that allow a better estimate of the environmental impacts generated by human activities.

Several environmental assessment studies applied to wine (Bosco et al., 2011; Bartocci et al., 2017; Petti, De Camillis, Raggi, & Vale, 2015) highlight the farming and packaging stages as those contributing the most to the total environmental impacts of wine. In this sense, organic farming is often proposed as a solution to mitigate the environmental effects caused by conventional farming (Seufert et al., 2012), which are mainly associated with a greater use of synthetic fertilisers and pesticides (Villanueva-Rey et al., 2014) and intensive tillage (Keesstra et al., 2018; Rodrigo-Comino et al., 2018). However, results tend to vary depending on the functional unit, and although the analyses per farm area usually show a greater impact of conventional agriculture, when taking the yield into account, the values of organic farming are higher in some impact categories (Meier et al., 2015).

Life cycle assessment (LCA) is a widely accepted methodology for evaluating the potential environmental impacts associated with the agri-food chain in general and with agricultural production systems in particular (Bosco et al., 2011; Schmidt Rivera et al., 2017). One of the main challenges when applying LCA to agricultural systems is that of modelling the emissions from fertiliser and pesticide application when performing the inventory analysis (Peña et al., 2019; Schmidt Rivera et al., 2017). These emissions are often estimated through models which consider generic emission factors (EFs). Specifically, the ones proposed in the IPCC (2006a) Tier 1 have been widely applied to estimate nitrogen emissions from fertilisers (e.g. Bacenetti et al., 2016; Ponstein, Meyer-aurich, & Prochnow, 2019; Ribal et al., 2017; Steenwerth et al., 2015), whereas the SALCA-P model (Prasuhn, 2006) is recommended by Nemecek et al. (2014) to estimate PO_4^{3-} emissions. In addition, the model proposed by Margni et al. (2002) is among the most widely used to calculate pesticide fate (e.g. Fusi et al., 2014; Neto, Diaz & Machado, 2013). However, other models take into account site-specific aspects, namely climate and soil characteristics. Among the most commonly used, both the one proposed by Brentrup et al. (2000) for fertiliser emissions and the PestLCI for pesticide fate can be highlighted (e.g. Bacenetti et al., 2015; Vázquez-Rowe et al., 2012; Villanueva-Rey et al., 2014).

Consequently, some studies have discussed the implications of choosing different nitrogen fertiliser and pesticide emission models in the LCA of agricultural products (e.g. Goglio et al., 2018; Peña et al., 2019; Perrin et al., 2014; Perrin et al., 2017; Peter et al., 2016; Schmidt Rivera et al., 2017). Likewise, Peña et al. (2019) developed a proposal to calculate pesticide fates which was contrasted with two other models, the one from Margni et al. (2002), which considers fixed share percentages, versus another one that not only takes into account the initial distribution (i.e., application method and crop characteristics) but also includes field emissions (Balsari et al., 2007; Felsot et al., 2010; Gil et al., 2014; Gil & Sinfort, 2005).

When comparing modelling approaches (MA) for emissions derived from fertiliser and pesticide application, the direct correlation implicit between the accuracy of the estimates and the effort made to obtain the information needed for the model is an important issue to be evaluated. It can be assumed that MA using site-specific information (S_{MA}) are more accurate in their estimates than those requiring generic information (G_{MA}). Hence, when environmental impacts are estimated considering S_{MA} and the results are significantly different from those estimated considering G_{MA}, the choice of S_{MA} is suggested, although greater efforts are required to obtain the model data (IPCC, 2006b). Conversely, if no significant differences are observed or in the absence of more accurate information, G_{MA} allow reliable estimates to be computed.

LCA has been applied to different Spanish wine PDOs, such as Conca de Barberà in Catalonia (Meneses et al., 2016), Ribeiro in Galicia (Vazquez-Rowe et al., 2012; Villanueva-Rey et al., 2014; 2018) or la Rioja (Gazulla et al., 2010; Flor et al., 2018). In addition, other studies have addressed different aspects related to winemaking in Spain (e.g. González-García et al., 2011; Rives et al., 2011). In order to produce new LCA-related results for the Spanish wine sector, this study aims to analyse the environmental profile of the most widespread viticultural practices in the Utiel-Requena PDO. In addition, since the influence of the estimation of fertiliser and pesticide emission models in vineyards has not been previously addressed, this study also aims to evaluate the differences between the environment impacts estimated considering G_{MA} (Baseline modelling or BM) versus S_{MA} (Alternative modelling or AM) in vineyards.

2. Materials and methods

This study was carried out applying the LCA methodology based on ISO standard guidelines (ISO, 2006a; 2006b; ISO, 2017) and using Gabi software v. 9.2.0.58 (Thinkstep, Leinfelden-Enchterdigen, Germany).

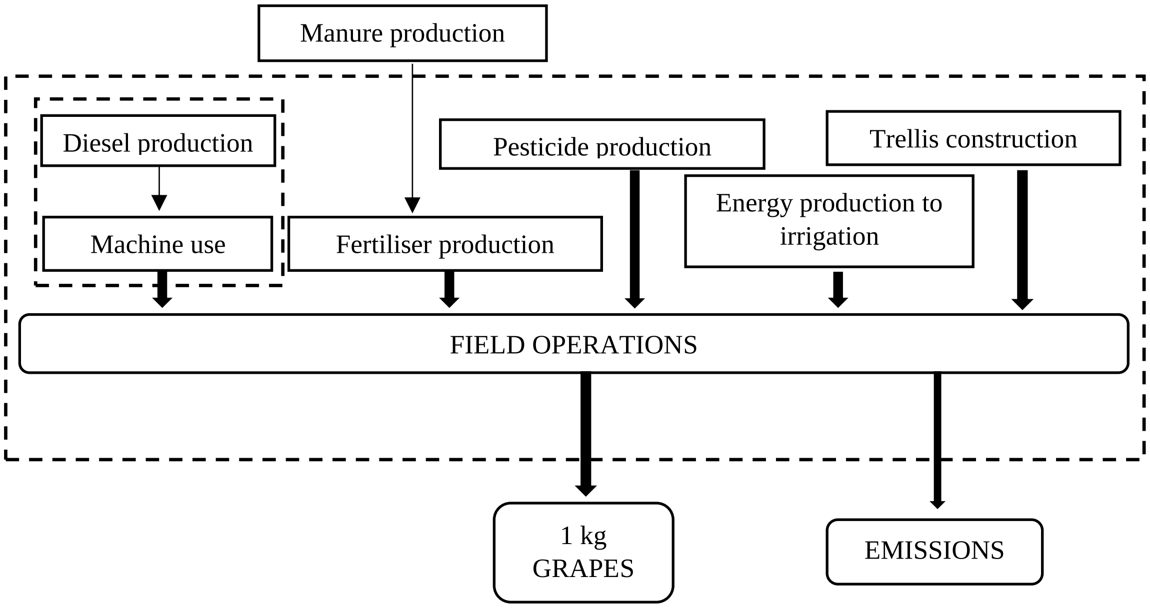
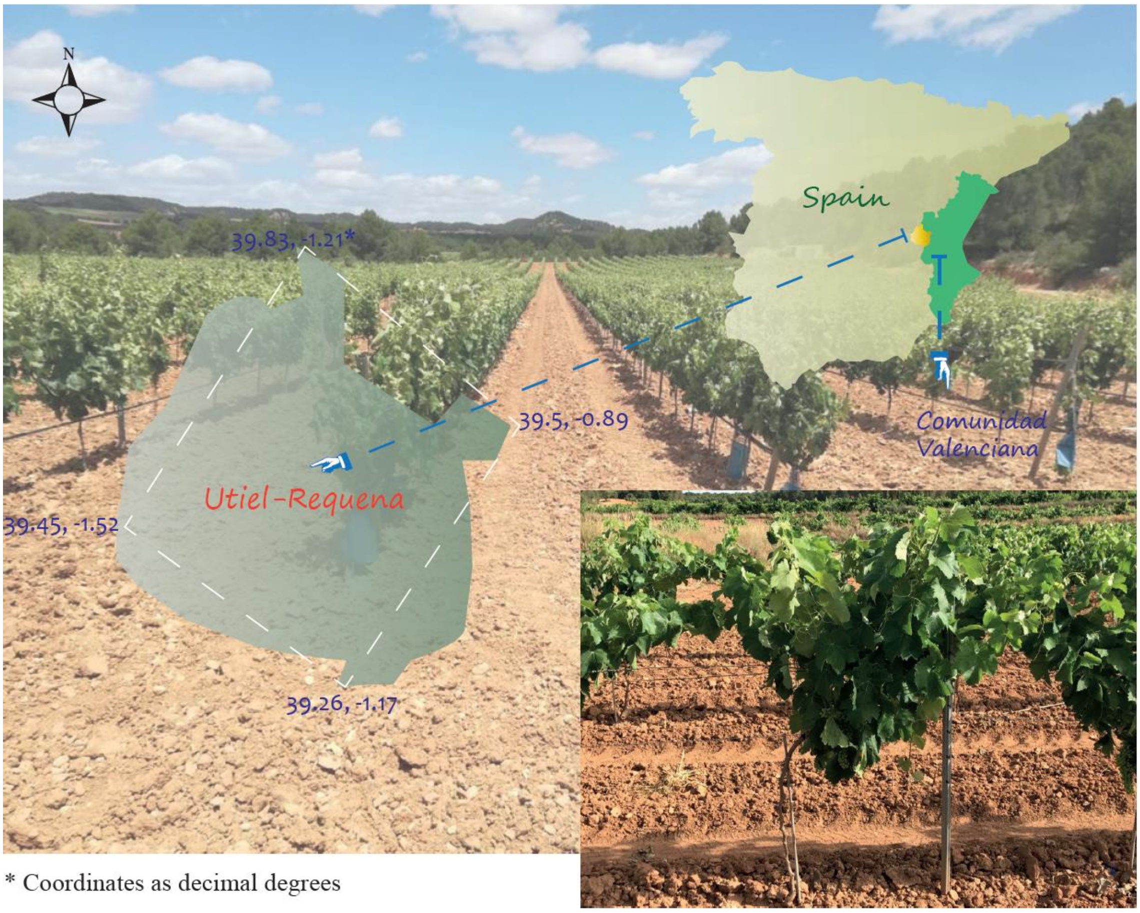


Fig. 1 System boundaries

2.1 Study area

Utiel-Requena is located in the west of the Valencian region (Figure 2) and comprising nine municipalities, it is 60-90 km from the Mediterranean and at 600 to 900 m above sea level. This region has a Mediterranean climate with continental features. Its average annual rainfall is 385 mm, with a wet period of 7 months (October to April), a semi-humid period of 2 months (May and September) and a dry period of 3 months (from June to August). The average temperature is about 14.6 ºC with a maximum of 20.6 ºC and a minimum of 8.6 ºC. As to the soil characteristics, it corresponds to Mediterranean red soils, of sedimentary origin, with limestone and siliceous characteristics and with a second horizon of clay accumulation is stand out (Buesa et al., 2017; IVIA, 2019).

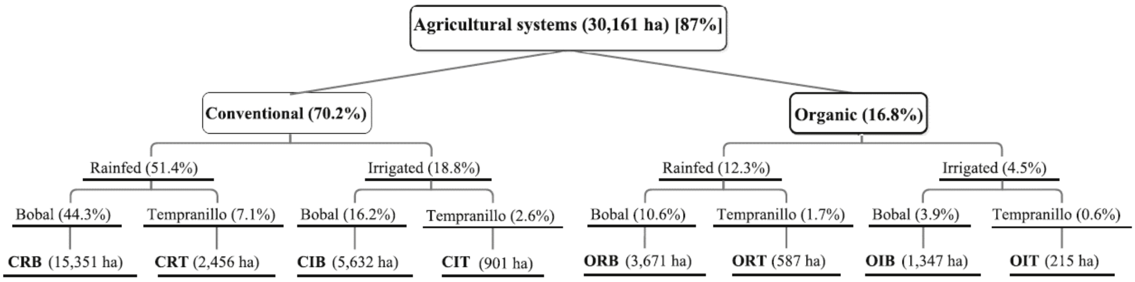


* Coordinates as decimal degrees

Fig. 2 Description of the area of study

2.2 Goal and scope

This LCA aims to carry out an environmental characterization of the most representative crop management systems in the production of wine grapes in the Utiel-Requena PDO and to evaluate the influence of fertiliser and pesticide emission modelling on the environmental profiles of the analysed systems. For this assessment, a season with standard agroclimatic conditions is considered.



CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety.

Fig. 3 Representative crop management systems of grape production in the Utiel-Requena PDO. In brackets, the area of each productive system is shown together with the percentage of area with respect to the total agricultural area in the PDO. (CAMACCDR, 2019b; Consejo Regulador de Utiel-Requena DO, 2019).

Following Villanueva-Rey et al. (2018), the functional unit (FU) is 1 kg of harvested grapes. System boundaries are set at the farm gate and the life cycle stages shown in Figure 1 have been taken into account. The system is structured from the most representative agricultural practices for wine grape production in the Utiel-Requena PDO and it includes both the emissions caused by the production of inputs and those derived from field operations, especially the use of fertilisers, pesticides and machinery.

Using direct interviews with technical staff of grape production cooperatives belonging to the PDO as a starting point, information on both the most common agricultural practices and the amount of inputs used was obtained. Although conventional farming was identified as the most common system in the PDO, organic farming is on the increase; in addition, within each system there are two types of technical management. The first one consists of goblet spur pruning without irrigation (gs-rainfed crop), while in the second one double guyot cane pruning with trellis is used and the crop is irrigated (dg-irrigated). Moreover, considering the main grape cultivars in the PDO (Bobal and Tempranillo), for the purposes of this study, eight representative productive systems have been configured (Figure 3). Figure 3 also shows the total vineyard surface area corresponding to the systems studied in the PDO, the area of each productive system together with the percentage of area of each one with respect to the total agricultural area in the PDO, estimated from official data (CAMACCDR, 2019b; *Consejo Regulador de Utiel-Requena DO, 2019*).

2.3 Life cycle inventory (LCI)

Tables 1 and 2 show the inputs and outputs, respectively, used in the environmental assessment of 1 kg of Bobal and Tempranillo grapes in the Utiel-Requena PDO for each productive system. In the subsequent sections, these data are detailed.

2.3.1 Agricultural field operations

Field operations in the gs-rainfed systems (CRB, CRT, ORB and ORT) include different activities, namely pruning, tillage, the application of fertilisers and pesticides, and harvesting. For the dg-irrigated systems (CIB, CIT, OIB and OIT), besides the activities included in the gs-rainfed ones, the trellis construction and irrigation activities are added.

2.3.2 Input production

The emissions from input production have been calculated using the processes from different databases; namely, Ecoinvent 3.5 (Wernet et al., 2016) for the Spanish electricity mix, pesticides, potassium 0-0-15 and ammonium sulphate, and Professional Gabi 8.7 for diesel, NPK 15-15-15 and galvanized steel production, and machinery use.

Table 1. LCI Inputs.

	CRB	CRT	CIB	CIT	ORB	ORT	OIB	OIT
Inputs kg of grapes^{-1}								
Pruning								
tractor use (h)	$1.4\cdot10^{-4}$	$1.7\cdot10^{-4}$	$4.4\cdot10^{-4}$	$5.7\cdot10^{-4}$	$1.8\cdot10^{-4}$	$2.2\cdot10^{-4}$	$5.3\cdot10^{-4}$	$7.3\cdot10^{-4}$
Tillage								
tractor use (h)	$1.3\cdot10^{-3}$	$1.5\cdot10^{-3}$	$8.3\cdot10^{-4}$	$1.1\cdot10^{-03}$	$1.6\cdot10^{-3}$	$2.0\cdot10^{-3}$	$1.0\cdot10^{-3}$	$1.4\cdot10^{-3}$

glyphosate (kg)			$5.0\cdot10^{-04}$	$6.4\cdot10^{-04}$				
Fertiliser application								
tractor use (h)	$1.4\cdot10^{-4}$	$1.7\cdot10^{-4}$	$1.1\cdot10^{-4}$	$1.4\cdot10^{-4}$	$1.2\cdot10^{-4}$	$1.5\cdot10^{-4}$	$8.9\cdot10^{-5}$	$1.2\cdot10^{-4}$
manure (kg)	$5.7\cdot10^{-1}$	$6.7\cdot10^{-1}$	$4.4\cdot10^{-1}$	$5.7\cdot10^{-1}$	$7.3\cdot10^{-1}$	$8.9\cdot10^{-1}$	$5.3\cdot10^{-1}$	$7.3\cdot10^{-1}$
NPK 15-15-15 (kg)	$1.2\cdot10^{-2}$	$1.4\cdot10^{-2}$						
ammonia sulphate (kg)			$2.2\cdot10^{-2}$	$2.9\cdot10^{-2}$				
potassium 0-0-15 (kg)			$4.7\cdot10^{-2}$	$6.1\cdot10^{-2}$				
Pesticide application								
tractor use (h)	$6.4\cdot10^{-4}$	$7.5\cdot10^{-4}$	$5.0\cdot10^{-4}$	$6.4\cdot10^{-4}$	$5.5\cdot10^{-4}$	$6.7\cdot10^{-4}$	$4.0\cdot10^{-4}$	$5.5\cdot10^{-4}$
copper oxychloride (kg)	$2.1\cdot10^{-3}$	$2.5\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.1\cdot10^{-3}$	$1.8\cdot10^{-3}$	$2.2\cdot10^{-3}$	$1.3\cdot10^{-3}$	$1.8\cdot10^{-3}$
sulphur (kg)	$8.6\cdot10^{-3}$	$1.0\cdot10^{-2}$	$6.7\cdot10^{-3}$	$8.6\cdot10^{-3}$	$7.3\cdot10^{-3}$	$8.9\cdot10^{-3}$	$5.3\cdot10^{-3}$	$7.3\cdot10^{-3}$
Irrigation								
water (l)			$7.9\cdot10^{1}$	$1.0\cdot10^{2}$			$9.5\cdot10^{1}$	$1.3\cdot10^{2}$
energy (MJ)			$7.9\cdot10^{-3}$	$1.0\cdot10^{-2}$			$9.5\cdot10^{-3}$	$1.3\cdot10^{-2}$
Trellis								
galvanized steel (kg)			$2.6\cdot10^{-1}$	$3.3\cdot10^{-1}$			$3.1\cdot10^{-1}$	$4.2\cdot10^{-1}$
Harvest								
tractor use (h)	$6.0\cdot10^{-5}$	$6.9\cdot10^{-5}$	$2.2\cdot10^{-5}$	$2.9\cdot10^{-5}$	$6.0\cdot10^{-5}$	$7.3\cdot10^{-5}$	$2.7\cdot10^{-5}$	$3.6\cdot10^{-5}$
harvester use (h)			$6.5\cdot10^{-5}$	$8.3\cdot10^{-5}$			$7.8\cdot10^{-5}$	$1.1\cdot10^{-4}$

CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety.

It must be pointed out that manure and tractor production are not considered. Sheep manure has a low economic value and its environmental burdens are allocated to other co-products derived from sheep farming. The tractor has a relatively long economic life; therefore, the loads associated with 1 kg grapes are not significant. As to the trellis construction, only the production of galvanized steel is included because it was identified as the only material with relative importance.

2.3.3 Emissions from fertiliser and pesticide application

The methodological approaches compared in this study follow different guidelines. Namely, in the BM, the IPCC (2006a) TIER 1 guidelines and SALCA-P model (Prasuhn, 2006) were used to estimate nitrogen emissions (direct and indirect N_2O, NH_3, NO_x and NO_3^-) and PO_4^{3-} emissions from fertilisers, respectively; whereas pesticide fate was estimated from Margni et al. (2002). On the other hand, in the AM, different modelling approaches were used for fertiliser emissions. Direct N_2O emissions were estimated according to the IPCC (2006a) TIER 2, using an EF for grape cultivation in the Mediterranean region from Cayuela et al. (2017), whereas indirect N_2O emissions were estimated following IPCC (2006a). As to NH_3, the TIER 2 EF of the European Environmental Agency guidelines (EMEP/EEA, 2019a) was used. For NO_x emissions, the Tier 1 EF from the same source was used, since no Tier 2 EF is proposed. Likewise, NO_3^- and PO_4^{3-} emissions were determined from nitrogen and phosphorus balances following MAPAMA (2018b;2018c), (See SM-2 and Table S2 and Table S3 for details).

Table 2. LCI Outputs.

	CRB	**CRT**	**CIB**	**CIT**	**ORB**	**ORT**	**OIB**	**OIT**
Outputs								
To the technosphere (kg ha^{-1})								
Products								
Grapes	7,000	6,000	9,000	7,000	5,500	4,500	7,500	5,500
To the environment kg grapes^{-1}								
Fertiliser emissions								
N_2O total_BM (kg)	$1.3\cdot10^{-4}$	$1.5\cdot10^{-4}$	$1.5\cdot10^{-4}$	$1.9\cdot10^{-4}$	$1.3\cdot10^{-4}$	$1.5\cdot10^{-4}$	$9.2\cdot10^{-5}$	$1.3\cdot10^{-4}$
N_2O total_AM (kg)	$4.2\cdot10^{-5}$	$5.9\cdot10^{-5}$	$1.1\cdot10^{-4}$	$1.6\cdot10^{-4}$	$4.2\cdot10^{-5}$	$6.0\cdot10^{-5}$	$5.1\cdot10^{-5}$	$6.9\cdot10^{-5}$
N_2O direct _BM (kg)	$1.1\cdot10^{-4}$	$1.2\cdot10^{-4}$	$1.3\cdot10^{-4}$	$1.7\cdot10^{-4}$	$1.0\cdot10^{-4}$	$1.2\cdot10^{-4}$	$7.5\cdot10^{-5}$	$1.0\cdot10^{-4}$

N_2O direct_AM (kg)	$3.1\cdot10^{-5}$	$3.6\cdot10^{-5}$	$6.7\cdot10^{-5}$	$8.7\cdot10^{-5}$	$3.1\cdot10^{-5}$	$3.8\cdot10^{-5}$	$4.3\cdot10^{-5}$	$5.9\cdot10^{-5}$
N_2O indirect_BM (kg)	$2.2\cdot10^{-5}$	$2.5\cdot10^{-5}$	$2.2\cdot10^{-5}$	$2.8\cdot10^{-5}$	$2.4\cdot10^{-5}$	$2.9\cdot10^{-5}$	$1.8\cdot10^{-5}$	$2.4\cdot10^{-5}$
N_2O indirect_AM (kg)	$1.1\cdot10^{-5}$	$2.2\cdot10^{-5}$	$3.8\cdot10^{-5}$	$7.0\cdot10^{-5}$	$1.0\cdot10^{-5}$	$2.2\cdot10^{-5}$	$7.7\cdot10^{-6}$	$1.0\cdot10^{-5}$
NH_3_BM (kg)	$8.7\cdot10^{-4}$	$1.0\cdot10^{-3}$	$8.8\cdot10^{-4}$	$1.1\cdot10^{-3}$	$9.7\cdot10^{-4}$	$1.2\cdot10^{-3}$	$7.1\cdot10^{-4}$	$9.7\cdot10^{-4}$
NH_3_AM (kg)	$6.7\cdot10^{-4}$	$7.8\cdot10^{-4}$	$1.2\cdot10^{-3}$	$1.5\cdot10^{-3}$	$6.4\cdot10^{-4}$	$7.8\cdot10^{-4}$	$4.7\cdot10^{-4}$	$6.4\cdot10^{-4}$
NO_X_BM (kg)	$1.4\cdot10^{-3}$	$1.6\cdot10^{-3}$	$1.4\cdot10^{-3}$	$1.8\cdot10^{-3}$	$1.5\cdot10^{-3}$	$1.9\cdot10^{-3}$	$1.1\cdot10^{-3}$	$1.5\cdot10^{-3}$
NO_X_AM (kg)	$3.0\cdot10^{-4}$	$3.5\cdot10^{-4}$	$3.4\cdot10^{-4}$	$4.4\cdot10^{-4}$	$3.0\cdot10^{-4}$	$3.7\cdot10^{-4}$	$2.2\cdot10^{-4}$	$3.0\cdot10^{-4}$
NO_3^-_BM (kg)	0	0	0	0	0	0	0	0
NO_3^-_AM (kg)	0	$3.6\cdot10^{-3}$	$7.5\cdot10^{-3}$	$1.8\cdot10^{-2}$	0	$3.5\cdot10^{-3}$	0	0
PO_4^{3-} _BM (kg)	$2.3\cdot10^{-3}$	$2.6\cdot10^{-3}$	$1.0\cdot10^{-3}$	$1.3\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.0\cdot10^{-3}$	$1.2\cdot10^{-3}$	$1.7\cdot10^{-3}$
PO_4^{3-} _AM (kg)	0	0	0	0	0	0	0	0
Pesticide emissions								
glyphosate to air_BM (kg)	0	0	$3.1\cdot10^{-5}$	$3.9\cdot10^{-5}$	0	0	0	0
glyphosate to agricultural soil_BM (kg)	0	0	$2.3\cdot10^{-4}$	$3.0\cdot10^{-4}$	0	0	0	0
glyphosate to fresh water_BM (kg)	0	0	$2.6\cdot10^{-5}$	$3.3\cdot10^{-5}$	0	0	0	0
glyphosate to air_AM (kg)	0	0	$1.5\cdot10^{-5}$	$2.0\cdot10^{-5}$	0	0	0	0
glyphosate to fresh water_AM (kg)	0	0	$2.6\cdot10^{-7}$	$3.3\cdot10^{-7}$	0	0	0	0
glyphosate to agricultural soil_AM (kg)	0	0	$8.4\cdot10^{-5}$	$1.1\cdot10^{-4}$	0	0	0	0
glyphosate to other soil_AM (kg)	0	0	$2.4\cdot10^{-5}$	$3.1\cdot10^{-5}$	0	0	0	0
copper oxychloride to air_BM (kg)	$7.5\cdot10^{-5}$	$8.8\cdot10^{-5}$	$5.8\cdot10^{-5}$	$7.5\cdot10^{-5}$	$6.4\cdot10^{-5}$	$7.8\cdot10^{-5}$	$4.7\cdot10^{-5}$	$6.4\cdot10^{-5}$
copper oxychloride to agricultural soil_BM (kg)	$5.7\cdot10^{-4}$	$6.7\cdot10^{-4}$	$4.5\cdot10^{-4}$	$5.7\cdot10^{-4}$	$4.9\cdot10^{-4}$	$6.0\cdot10^{-4}$	$3.6\cdot10^{-4}$	$4.9\cdot10^{-4}$
copper oxychloride to fresh water_BM (kg)	$6.4\cdot10^{-5}$	$7.4\cdot10^{-5}$	$5.0\cdot10^{-5}$	$6.4\cdot10^{-5}$	$5.4\cdot10^{-5}$	$6.6\cdot10^{-5}$	$4.0\cdot10^{-5}$	$5.4\cdot10^{-5}$
copper oxychloride to air_AM (kg)	$7.5\cdot10^{-6}$	$8.7\cdot10^{-6}$	$5.8\cdot10^{-6}$	$7.5\cdot10^{-6}$	$6.4\cdot10^{-6}$	$7.8\cdot10^{-6}$	$4.7\cdot10^{-6}$	$6.4\cdot10^{-6}$
copper oxychloride to fresh water_AM (kg)	$6.4\cdot10^{-7}$	$7.4\cdot10^{-7}$	$5.0\cdot10^{-7}$	$6.4\cdot10^{-7}$	$5.4\cdot10^{-7}$	$6.6\cdot10^{-7}$	$4.0\cdot10^{-7}$	$5.4\cdot10^{-7}$
copper oxychloride to agricultural soil_AM (kg)	$2.5\cdot10^{-4}$	$3.0\cdot10^{-4}$	$1.7\cdot10^{-4}$	$2.1\cdot10^{-4}$	$2.2\cdot10^{-4}$	$2.6\cdot10^{-4}$	$1.3\cdot10^{-4}$	$1.8\cdot10^{-4}$
copper oxychloride to other soil_AM (kg)	$6.2\cdot10^{-5}$	$7.3\cdot10^{-5}$	$4.9\cdot10^{-5}$	$6.2\cdot10^{-5}$	$5.3\cdot10^{-5}$	$6.5\cdot10^{-5}$	$3.9\cdot10^{-5}$	$5.3\cdot10^{-5}$
sulphur to air_BM (kg)	$8.5\cdot10^{-4}$	$9.9\cdot10^{-4}$	$6.6\cdot10^{-4}$	$8.5\cdot10^{-4}$	$7.2\cdot10^{-4}$	$8.8\cdot10^{-4}$	$5.3\cdot10^{-4}$	$7.2\cdot10^{-4}$
sulphur to agricultural soil_BM (kg)	$6.5\cdot10^{-3}$	$7.6\cdot10^{-3}$	$5.0\cdot10^{-3}$	$6.5\cdot10^{-3}$	$5.5\cdot10^{-3}$	$6.7\cdot10^{-3}$	$4.0\cdot10^{-3}$	$5.5\cdot10^{-3}$
sulphur to fresh water_BM (kg)	$7.2\cdot10^{-4}$	$8.4\cdot10^{-4}$	$5.6\cdot10^{-4}$	$7.2\cdot10^{-4}$	$6.1\cdot10^{-4}$	$7.5\cdot10^{-4}$	$4.5\cdot10^{-4}$	$6.1\cdot10^{-4}$
sulphur to air_AM (kg)	$1.3\cdot10^{-3}$	$1.5\cdot10^{-3}$	$9.9\cdot10^{-4}$	$1.3\cdot10^{-3}$	$1.1\cdot10^{-3}$	$1.3\cdot10^{-3}$	$7.9\cdot10^{-4}$	$1.1\cdot10^{-3}$
sulphur to fresh water_AM (kg)	$7.2\cdot10^{-6}$	$8.4\cdot10^{-6}$	$5.6\cdot10^{-6}$	$7.2\cdot10^{-6}$	$6.1\cdot10^{-6}$	$7.5\cdot10^{-6}$	$4.5\cdot10^{-6}$	$6.1\cdot10^{-6}$
sulphur to agricultural soil_AM (kg)	$2.5\cdot10^{-3}$	$2.9\cdot10^{-3}$	$1.6\cdot10^{-3}$	$2.1\cdot10^{-3}$	$2.1\cdot10^{-3}$	$2.6\cdot10^{-3}$	$1.3\cdot10^{-3}$	$1.8\cdot10^{-3}$
sulphur to other soil_AM (kg)	$6.1\cdot10^{-4}$	$7.1\cdot10^{-4}$	$4.7\cdot10^{-4}$	$6.1\cdot10^{-4}$	$5.1\cdot10^{-4}$	$6.3\cdot10^{-4}$	$3.8\cdot10^{-4}$	$5.1\cdot10^{-4}$

BM: baseline modelling; AM: alternative modelling; CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety.

Table 2 shows the on-field emissions for each production system and methodological approach. It must be noted that NO_3^- emissions are zero with BM because, according to the IPCC (2006a), if the difference between the rainfall during the rainy season and the potential evaporation in the same period is lower than the soil water holding capacity and drip irrigation is carried out, the leaching fraction is zero. On the other hand, PO_4^{3-} emissions are zero when using AM, whereas NO_3^- are also zero in CRB, ORB, OIB and OIT, because in these cases the phosphorus and nitrogen balances were negative (see S2 and S3). Finally, it is worth mentioning that, in general, the emissions from conventional irrigation models tend to be higher than those from the other productive systems. However, some heterogeneity is found which is further analysed along with the characterization of the environmental impacts.

When estimating pesticide emissions according to Peña et al. (2019), data from different sources were used. On the one hand, the leaf area index (LAI) was obtained from Pérez Bartolomé (2002) taking the simple average of the LAI for June over three consecutive years. The capture coefficient (Kp) was 0.55 (Peña et al., 2019), while the water-to-soil area ratio of 0.1 was obtained from Juraske & Sanjuán (2011). Finally, following Balsari et al. (2007) and with data from the Julius Kühn Institute (JKI, 2019), the drift percentage was set at 17% with a 50% reduction for the use of anti-drift nozzles; therefore, the drift percentage remained at 8.5%.

2.3.4 Blue water consumption from irrigation

Following AWARE guidelines (Pieper, Kupfer, Thylmann, & Bos, 2018), blue water consumption was estimated through the crop evapotranspiration by using equation (1):

$$ET_c = K_c \cdot ET_o \qquad (1)$$

Where, ET_c is the crop evapotranspiration ($mm \cdot day^{-1}$), K_c is the crop coefficient (dimensionless), and ET_o is the reference crop evapotranspiration ($mm \cdot d^{-1}$). Both ET_o and K_C were obtained as the average of the data for the Requena municipality published by IVIA (2019) corresponding to the last ten years (2009-2018). In SM-1, detailed estimates of blue water consumption are shown.

2.4 Impact categories and impact assessment methods

The impact categories normally analysed in LCAs were calculated in this study, namely: climate change (CC) as CO_2-eq. for a time horizon of 100 years (kg); fine particulate matter formation (FPMF) as kg PM2.5 eq.; fossil (FD) and metal depletion (MD) as kg Cu eq.; freshwater eutrophication (FwE) as kg P eq.; marine eutrophication (ME) as kg N eq.; terrestrial acidification (TA) as kg SO_2 eq.; photochemical ozone formation, ecosystems (POFe) as kg NOx eq.; photochemical ozone formation, human health (POFh) as kg NOx eq.; stratospheric ozone depletion (SOD) as kg CFC-11 eq.; land use (LU) as Annual crop eq.·y; ionizing radiation (IR) as Bq C-60 eq. to air; water scarcity (WS) as m^3 world equiv.; freshwater ecotoxicity (ET) as CTUe; both cancer (HTc) and non-cancer (HTnc) human toxicity as CTUh. The toxicity related impact categories were characterized through UseTox 2.3 (Hauschild et al., 2008; Rosenbaum et al., 2008), the water scarcity category with AWARE (Boulay et al., 2018) and for the remaining categories, the ReCiPe 2016 v1.1 method was used (Huijbregts et al., 2017).

It should be mentioned that for the HTnc and ET, interim characterization factors were used to compute the effects of on-field pesticide emissions, as there are no recommended characterization factors for copper-based pesticides. It should also be noted that there are no characterization factors available for sulphur-based pesticides; hence, their toxic consequences were not taken into account.

2.5 Statistical analysis

The interpretation of the results is carried out from both descriptive and inferential analyses. The descriptive analysis allows a first approximation to identify the relative contribution of the different sources of emissions or the consumption of resources to each impact category and suggest possible differences between the results of the productive systems and methodological approaches analysed.

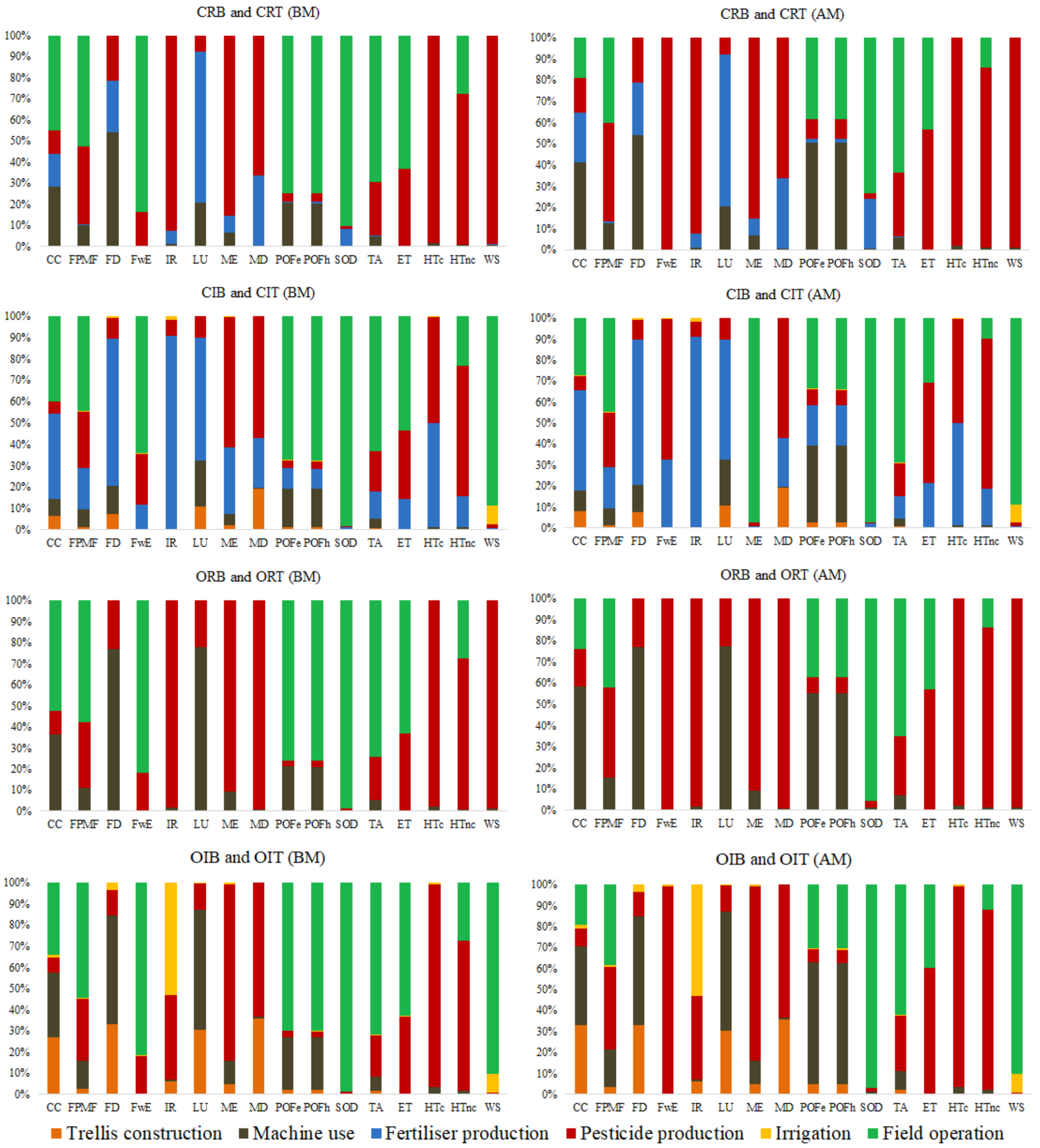
The inferential analysis seeks to assess whether the differences identified in the descriptive analysis are statistically significant or not, to this end the IBM SPSS statistics software v25 was used. The Mann-Whiney U test was identified as the most appropriate technique for the development of the inferential comparisons. This is due to the fact that when applying Shapiro-Wilk and Kolmogorov-Smirnov tests to small samples (n <30) and large samples (n> 30), respectively, no normality in the distributions of each dependent variable analysed was found at a 5% significance level; in addition, it has been considered that there is no interdependence among the classification variables. In this study, the sample size is determined by combining the productive systems with the methodological approaches to estimate the emissions from fertilisers and pesticides. A 5% significance level is used; hence, when Mann-Whiney U's p-value is lower than 5%, it means that there are significant differences between the compared results. Detailed information about the statistical methodology applied in this study can be found in MacFarland & Yates (2016).

3. Results

3.1 Productive systems

Table 3 shows the results of the environmental impacts for each productive system using the AM and BM emissions estimation, while Fig. 4 shows the contribution analysis. It can be observed how field operations, together with the production of fertilisers and pesticides, are the main sources of environmental impacts in the conventional systems. In the context of the organic system, field operations, together with the use of machinery and pesticide production, are the main contributors to most of the impact categories. Nevertheless, in some productive systems, such as organic dg-irrigated systems (OIB, OIT), both irrigation and trellis construction take on importance for some environmental impact categories (CC, FD, LU, MD and IR).

The high relative contribution that on-field emissions from fertilisers and pesticides (field operation) make to many impact categories (CC, FPMF, FwE, ME, POFh, POFe, SOD, TA, ET and HTnc) underlines the fact that the model used to estimate these emissions can modify the relative contribution of the life cycle stages to the total impact.



BM: baseline modelling; AM: alternative modelling; CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; CC: Climate change; FPMF: Fine particulate matter formation; FD: Fossil depletion; FeW: Freshwater eutrophication; IR: Ionizing radiation; LU: Land use; ME: Marine eutrophication; MD: Metal depletion; POFe: Photochemical ozone formation, ecosystems; POFh: Photochemical ozone formation, human health; SOD: Stratospheric ozone depletion; TA: Terrestrial acidification; ET: Freshwater ecotoxicity; HTc: Human toxicity, cancer; HTnc: Human toxicity, non-cancer; WS: Water scarcity.

Fig. 4 Relative contribution of life cycle stages to the environmental impacts of the productive systems

A first analysis of the environmental impacts obtained for the production systems being analysed (Table 3) indicates that, generally speaking, the environment impacts are higher for the Tempranillo cultivar than for the Bobal in every category. This is because, for a fixed amount of applied inputs, the yield of the Tempranillo grape cultivar is lower and, consequently, the environmental impacts generated per kg grapes are greater than those for the Bobal cultivar.

As can be observed in Table 3, excepting the WS category, the heaviest pollutants are the conventional dg-irrigated systems (CIB and CIT) due to the fact that more synthetic fertilisers (specifically, ammonium sulphate and potassium 0-0-15) are used in these systems than in the others. Although potassium fertiliser is not associated with on-field emissions, the impacts related to its production contribute to the differences observed. As to the WS, organic dg-irrigated systems (OIB, OIT) are the ones that generate the greatest impact; this is because, despite the amount of water per hectare used being the same in every irrigated system, the organic dg-irrigated systems (OIB and OIT) are the ones with the lowest yield.

Table 3 Results of the environmental impacts of grape production in the Utiel-Requena PDO. FU: 1 kg grapes.

Impact Categories	**CRB**	**CRT**	**CIB**	**CIT**	**ORB**	**ORT**	**OIB**	**OIT**
Climate change [kg CO_2 eq.] BM	$9.3\cdot10^{-2}$	$1.1\cdot10^{-1}$	$2.5\cdot10^{-1}$	$3.2\cdot10^{-1}$	$7.8\cdot10^{-2}$	$9.6\cdot10^{-2}$	$9.2\cdot10^{-2}$	$1.3\cdot10^{-1}$
Climate change [kg CO_2 eq.] AM	$6.7\cdot10^{-2}$	$8.1\cdot10^{-2}$	$2.3\cdot10^{-1}$	$3.1\cdot10^{-1}$	$5.3\cdot10^{-2}$	$6.8\cdot10^{-2}$	$8.0\cdot10^{-2}$	$1.1\cdot10^{-1}$
Fine particulate matter formation [kg PM2.5 eq.] BM	$7.0\cdot10^{-4}$	$8.2\cdot10^{-4}$	$8.5\cdot10^{-4}$	$1.1\cdot10^{-3}$	$7.1\cdot10^{-4}$	$8.7\cdot10^{-4}$	$5.6\cdot10^{-4}$	$7.6\cdot10^{-4}$
Fine particulate matter formation [kg PM2.5 eq.] AM	$5.4\cdot10^{-4}$	$6.2\cdot10^{-4}$	$8.0\cdot10^{-4}$	$1.0\cdot10^{-3}$	$4.9\cdot10^{-4}$	$6.0\cdot10^{-4}$	$4.0\cdot10^{-4}$	$5.4\cdot10^{-4}$
Freshwater eutrophication [kg P eq.] BM	$8.9\cdot10^{-4}$	$1.0\cdot10^{-3}$	$5.2\cdot10^{-4}$	$6.7\cdot10^{-4}$	$6.7\cdot10^{-4}$	$8.2\cdot10^{-4}$	$4.9\cdot10^{-4}$	$6.7\cdot10^{-4}$
Freshwater eutrophication [kg P eq.] AM	$1.4\cdot10^{-4}$	$1.7\cdot10^{-4}$	$1.9\cdot10^{-4}$	$2.4\cdot10^{-4}$	$1.2\cdot10^{-4}$	$1.5\cdot10^{-4}$	$9.1\cdot10^{-5}$	$1.2\cdot10^{-4}$
Marine eutrophication [kg N eq.] BM	$1.1\cdot10^{-5}$	$1.3\cdot10^{-5}$	$1.3\cdot10^{-5}$	$1.7\cdot10^{-5}$	$8.8\cdot10^{-6}$	$1.1\cdot10^{-5}$	$7.0\cdot10^{-6}$	$9.5\cdot10^{-6}$
Marine eutrophication [kg N eq.] AM	$1.1\cdot10^{-5}$	$2.6\cdot10^{-4}$	$5.4\cdot10^{-4}$	$1.3\cdot10^{-3}$	$8.8\cdot10^{-6}$	$2.5\cdot10^{-4}$	$7.0\cdot10^{-6}$	$9.5\cdot10^{-6}$
Photochemical ozone formation, ecosystems [kg NOx eq.] BM	$1.9\cdot10^{-3}$	$2.2\cdot10^{-3}$	$2.2\cdot10^{-3}$	$2.8\cdot10^{-3}$	$2.1\cdot10^{-3}$	$2.6\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.3\cdot10^{-3}$
Photochemical ozone formation, Ecosystems [kg NOx eq.] AM	$8.2\cdot10^{-4}$	$9.6\cdot10^{-4}$	$1.1\cdot10^{-3}$	$1.4\cdot10^{-3}$	$8.4\cdot10^{-4}$	$1.0\cdot10^{-3}$	$7.6\cdot10^{-4}$	$1.0\cdot10^{-3}$
Photochemical ozone formation, human health [kg NOx eq.] BM	$1.9\cdot10^{-3}$	$2.2\cdot10^{-3}$	$2.2\cdot10^{-3}$	$2.8\cdot10^{-3}$	$2.1\cdot10^{-3}$	$2.5\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.3\cdot10^{-3}$
Photochemical ozone formation, human health [kg NOx eq.] AM	$8.2\cdot10^{-4}$	$9.5\cdot10^{-4}$	$1.1\cdot10^{-3}$	$1.4\cdot10^{-3}$	$8.4\cdot10^{-4}$	$1.0\cdot10^{-3}$	$7.6\cdot10^{-4}$	$1.0\cdot10^{-3}$
Stratospheric ozone depletion [kg CFC-11 eq.] BM	$1.6\cdot10^{-6}$	$1.8\cdot10^{-6}$	$1.7\cdot10^{-6}$	$2.2\cdot10^{-6}$	$1.4\cdot10^{-6}$	$1.7\cdot10^{-6}$	$1.0\cdot10^{-6}$	$1.4\cdot10^{-6}$
Stratospheric ozone depletion [kg CFC-11 eq.] AM	$6.4\cdot10^{-7}$	$8.5\cdot10^{-7}$	$1.2\cdot10^{-6}$	$1.8\cdot10^{-6}$	$4.8\cdot10^{-7}$	$6.9\cdot10^{-7}$	$5.8\cdot10^{-7}$	$7.9\cdot10^{-7}$
Terrestrial acidification [kg SO_2 eq.] BM	$3.2\cdot10^{-3}$	$3.7\cdot10^{-3}$	$3.6\cdot10^{-3}$	$4.6\cdot10^{-3}$	$3.3\cdot10^{-3}$	$4.1\cdot10^{-3}$	$2.5\cdot10^{-3}$	$3.5\cdot10^{-3}$
Terrestrial acidification [kg SO_2 eq.] AM	$2.4\cdot10^{-3}$	$2.8\cdot10^{-3}$	$3.8\cdot10^{-3}$	$4.9\cdot10^{-3}$	$2.2\cdot10^{-3}$	$2.7\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.4\cdot10^{-3}$
Human toxicity, non-canc. [CTUh] BM	$2.5\cdot10^{-7}$	$2.9\cdot10^{-7}$	$2.3\cdot10^{-7}$	$2.9\cdot10^{-7}$	$2.1\cdot10^{-7}$	$2.6\cdot10^{-7}$	$1.5\cdot10^{-7}$	$2.1\cdot10^{-7}$
Human toxicity, non-canc. [CTUh] AM	$2.1\cdot10^{-7}$	$2.4\cdot10^{-7}$	$2.0\cdot10^{-7}$	$2.5\cdot10^{-7}$	$1.8\cdot10^{-7}$	$2.2\cdot10^{-7}$	$1.3\cdot10^{-7}$	$1.7\cdot10^{-7}$
Freshwater ecotoxicity [CTUe] BM	$6.2\cdot10^{3}$	$7.2\cdot10^{3}$	$5.7\cdot10^{3}$	$7.3\cdot10^{3}$	$5.3\cdot10^{3}$	$6.4\cdot10^{3}$	$3.9\cdot10^{3}$	$5.3\cdot10^{3}$
Freshwater ecotoxicity [CTUe] AM	$4.0\cdot10^{3}$	$4.7\cdot10^{3}$	$3.8\cdot10^{3}$	$4.9\cdot10^{3}$	$3.4\cdot10^{3}$	$4.2\cdot10^{3}$	$2.4\cdot10^{3}$	$3.2\cdot10^{3}$
Fossil depletion [kg oil eq.]	$1.6\cdot10^{-2}$	$1.9\cdot10^{-2}$	$6.6\cdot10^{-2}$	$8.5\cdot10^{-2}$	$1.3\cdot10^{-2}$	$1.6\cdot10^{-2}$	$1.8\cdot10^{-2}$	$2.5\cdot10^{-2}$
Ionizing radiation [Bq C-60 eq. to air]	$8.9\cdot10^{-4}$	$1.0\cdot10^{-3}$	$3.5\cdot10^{-2}$	$4.5\cdot10^{-2}$	$7.1\cdot10^{-4}$	$8.7\cdot10^{-4}$	$1.3\cdot10^{-3}$	$1.8\cdot10^{-3}$
Land use [Annual crop eq.·y]	$9.0\cdot10^{-3}$	$1.0\cdot10^{-2}$	$8.2\cdot10^{-3}$	$1.1\cdot10^{-2}$	$2.7\cdot10^{-3}$	$3.2\cdot10^{-3}$	$3.5\cdot10^{-3}$	$4.7\cdot10^{-3}$
Metal depletion [kg Cu eq.]	$3.1\cdot10^{-3}$	$3.6\cdot10^{-3}$	$3.1\cdot10^{-3}$	$4.0\cdot10^{-3}$	$1.7\cdot10^{-3}$	$2.1\cdot10^{-3}$	$2.0\cdot10^{-3}$	$2.7\cdot10^{-3}$
Human toxicity, cancer [CTUh]	$9.1\cdot10^{-9}$	$1.1\cdot10^{-8}$	$1.5\cdot10^{-8}$	$1.9\cdot10^{-8}$	$7.8\cdot10^{-9}$	$9.5\cdot10^{-9}$	$5.8\cdot10^{-9}$	$7.9\cdot10^{-9}$
Water scarcity [m^3 world equiv.]	$6.4\cdot10^{-2}$	$7.5\cdot10^{-2}$	$3.5\cdot10^{0}$	$6.6\cdot10^{0}$	$5.4\cdot10^{-2}$	$6.7\cdot10^{-2}$	$4.2\cdot10^{0}$	$8.3\cdot10^{0}$

BM: baseline modelling; AM: alternative modelling; CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety.

Likewise, the impacts of conventional gs-rainfed systems (CRB, CRT) are greater than those of their organic peers (ORB, ORT) in every impact category, except POFh, POFe, FPMF and TA. These differences are associated with the production of pesticides, which are applied in greater quantity in conventional gs-rainfed systems, and also with the production of NPK 15-15-15, which is the only synthetic fertiliser applied in those conventional systems. In addition, the POFh and POFe values for organic gs-rainfed systems (ORB, ORT) are also higher with respect to the conventional gs-rainfed ones

(CRB, CRT); this can be attributed to the higher NOx emissions in the organic gs-rainfed systems brought about by the lower yield.

As regards the FPMF and TA categories, comparisons between the gs-rainfed systems (CRT, CRB vs. ORT, ORB) evidence sensitivity to the modelling of NH_3 emissions. Table 3 shows that when applying the BM approach, the results in these categories are favourable to those obtained the conventional gs-rainfed systems (CRB, CRT); conversely, when using the AM approach, opposite results are obtained. This is because NH_3 emissions per kg grapes are greater in the organic gs-rainfed systems and are overestimated when using the BM approach.

Table 4. Assessment of differences between productive factors per impact category analysed. Mann-Whiney U test, 5% significance level.

Impact Categories	Tempranillo vs. Bobal	Conventional gs-rainfed vs. Organic gs-rainfed	Conventional dg-irrigated vs. Organic dg-irrigated
Climate change	*	*	*
Fine particulate matter formation	*	nd	*
Fossil depletion	*	*	*
Freshwater eutrophication	*	*	*
Ionizing radiation	*	*	*
Land use	*	*	*
Marine eutrophication	*	*	*
Metal depletion	*	*	*
Photochemical ozone formation, ecosystems	*	**	*
Photochemical ozone formation, human health	*	**	*
Stratospheric ozone depletion	*	*	*
Terrestrial acidification	*	nd	*
Freshwater ecotoxicity	*	*	*
Human toxicity, cancer	*	*	*
Human toxicity, non-cancer	*	*	*
Water scarcity	*	*	**
	* Tempranillo > Bobal	* Conventional gs-rainfed > Organic gs-rainfed ** Organic gs-rainfed > Conventional gs-rainfed	* Conventional dg-irrigated > Organic dg-irrigated ** Organic dg-irrigated > Conventional dg-irrigated
nd: no differences			

Table 4 shows that all of these differences detailed in the above paragraphs are statistically significant, except in FPMF and TA in the gs-rainfed systems, which, as already mentioned, obtain different results depending on the modelling approach for NH_3 emissions.

Table 5. Assessment of differences between productive systems per impact category analysed. Mann-Whiney U test, 5% significance level.

	CRB	CRT	CIB	CIT	ORB	ORT	OIB	OIT
Climate change				**	*			
Fine particulate matter formation				**			*	
Fossil depletion				**	*			

Freshwater eutrophication		**		**			*	
Ionizing radiation				**	*			
Land use				**	*			
Marine eutrophication				**			*	
Metal depletion				**	*			
Photochemical ozone formation, Ecosystems				**			*	
Photochemical ozone formation, human health				**			*	
Stratospheric ozone depletion				**	*		*	
Terrestrial acidification				**			*	
Freshwater ecotoxicity				**			*	
Human toxicity, cancer				**			*	
Human toxicity, non-cancer				**			*	
Water scarcity					*			**

* Lowest impact; ** Greatest impact.
CRB: conventional system, with goblet spur pruning, rainfed, Bobal variety; CRT: conventional system, with goblet spur pruning, rainfed, Tempranillo variety; CIB: conventional system, double guyot cane pruning with trellis, irrigated, Bobal variety; CIT: conventional system, double guyot cane pruning with trellis, irrigated, Tempranillo variety; ORB: organic system, with goblet spur pruning, rainfed, Bobal variety; ORT: organic system, with goblet spur pruning, rainfed, Tempranillo variety; OIB: organic system, double guyot cane pruning with trellis, irrigated, Bobal variety; OIT: organic system, double guyot cane pruning with trellis, irrigated, Tempranillo variety.

To sum up , the results in Table 5 suggest that of the analysed production systems, ORB is significantly more environmentally viable for CC, FD, IR, LU, MD and WS categories, whereas OIB is better for FPMF, FwE, ME, POFe, POFh TA, ET, HTc and HTnc categories. As to SOD, both OIB and ORB exhibit the lowest impact. Even though neither of these two systems present the best environmental profile individually, in none of the impact categories analysed do they have the worst environmental position. In this same vein, the worst environmental profile corresponds to the CIT system for every impact category except WS, where OIT is the worst. It must be highlighted that in the case of the FwE, no significant differences are found between CIT and CRT systems, thus sharing the worst position.

3.2 Emission modelling approaches

As to the modelling approach, the results of on-field fertiliser and pesticide emissions tend to be lower when applying AM than with BM; thus, for the impact categories in which those emissions have an effect, that is, CC, FPMF, FwE, POFe, POFh, SOD, TA, HTnc and ET, the results also lower (Table 3). However, it is worth mentioning that the NH_3 estimates for the conventional dg-irrigated systems (CIB and CIT) are higher with AM than with BM, making FPMF and TA higher in these production systems. In addition, as NO_3^- emissions for all the production systems are zero when using BM, ME is higher with AM than BM; however, when using AM, NO_3^- emissions are also zero for CRB, ORB, OIB and OIT production systems and no differences are observed for ME.

Table 6. Assessment of differences in modelling proposals for calculation of on-field emissions. Mann-Whiney U test, 5% significance level.

Impact Categories	AM vs BM	N_2O	NH_3	NO_x	NO_3^-	PO_4^{3-}	Pesticides fate
CC	nd	*	nd	nd	nd	nd	nd
FPMF	*	nd	nd	*	nd	nd	nd

FwE	*	nd	nd	nd	nd	*	nd
ME	nd	nd	nd	nd	nd	nd	nd
POFe	*	nd	nd	*	nd	nd	nd
POFh	*	nd	nd	*	nd	nd	nd
SOD	*	*	nd	nd	nd	nd	nd
TA	*a	nd	nd	*	nd	nd	nd
ET	*	nd	nd	nd	nd	nd	*
HTnc	*a	nd	nd	nd	nd	nd	*
	* BM>AM	* IPCC (2006a) Tier 1 > Tier 2		* IPCC (2006a) Tier 1 > EEA (2019a) Tier 1		* SALCA-P. >P Balance	* Margni et al. (2002) > Peña et al. (2019)

*a significance to 10%; nd: no significant differences found; BM: baseline modelling; AM: alternative modelling; CC: Climate change; FPMF: Fine particulate matter formation; FeW: Freshwater eutrophication; ME: Marine eutrophication; POFe: Photochemical ozone formation, ecosystems; POFh: Photochemical ozone formation, human health; SOD: Stratospheric ozone depletion; TA: Terrestrial acidification; ET: Freshwater ecotoxicity; HTnc: Human toxicity, non-canc.

Table 6 shows the significance of the differences between the methodological approaches to the estimation of on-field emissions. As expected, each emission-modelling approach is significant for those impact categories with which it is involved. That is, the modelling of N_2O emissions is significant for CC and SOD; NO_x modelling for PFMF, POFe, POFh and TA; PO_4^{3-} modelling for FwE and the modelling of pesticide emissions for HTnc and ET. The significant differences found suggest an overestimation of N_2O, NO_x, PO_4^{3-} and pesticide emissions when applying the BM approach with respect to those obtained with the AM approach. It should also be noted that, in the case of NH_3 and NO_3^- emissions, no significant differences were found between the two methodological approaches.

Analysing the methodological approaches used (AM vs. BM) in an integral way (Table 6, first column), significant differences may be observed in the FPMF, FwE, POFe, POFh, SOD, TA, ET and HTnc impact categories. This indicates that the application of the BM approach instead of the AM leads to an overestimation of the results in every case. It is noteworthy that for the TA and THnc impact categories the test was validated at 10% significance level, which is a widely accepted level for hypothesis tests, together with 5% and 1%.

3.3 Uncertainty Analysis

Models are a simplification of real systems and hence they are not exact, thus they inherently hold uncertainty. In this regard, modelling with specific data reduces uncertainty against generic modelling, since it reduces bias and better represents the complexity of the system under analysis (IPCC, 2006b). In this sense, it can be argued that the AM approach is more accurate and therefore with less uncertainty compared to the BM approach. However, it is possible that in the quantification of the uncertainty cases may occur in which the results show that the uncertainty of AM is greater than in BM; this is due to the incomplete quantification of the uncertainty either because of computational complexity or because of lack of information (IPCC, 2006b).

Table 7. Inputs parameters for uncertainty analysis

Parameter	Unit	Baseline	Min	Max	Source
Indirect N_2O from (NH_3+NO_X)	kg N_2O–N (kg NH_3–N + NO_X–N volatilised)$^{-1}$	1.00%	0.20%	5.00%	IPCC (2006a)
Indirect N_2O from (NO_3-)	kg N_2O–N (kg N leaching/runoff)$^{-1}$	0.75%	0.05%	2.50%	IPCC (2006a)

BM					
Direct N_2O (EF)	kg N_2O-N kg N^{-1}	1.00%	0.30%	3.00%	IPCC (2006a)
NH_3 + NO_X (synthetic fertilisers)	(kg NH_3–N + NO_X–N) kg N^{-1}	10.00%	3.00%	30.00%	IPCC (2006a)
NH_3 + NO_X (Organic fertilisers)	(kg NH_3–N + NO_X–N) kg N^{-1}	20.00%	5.00%	50.00%	IPCC (2006a)
AM					
Direct N_2O rainfed (EF)	kg N_2O-N kg N^{-1}	0.27%	0.06%	0.48%	Cayuela et al. (2017)
Direct N_2O irrigated	kg N_2O-N kg N^{-1}	0.51%	0.25%	0.77%	Cayuela et al. (2017)
NO_X	Kg NO_X kg N^{-1}	4.00%	0.50%	10.40%	EEA (2019a)
NH_3_npk15	Kg NH_3 kg N^{-1}	9.40%	1.82%	23.50%	EEA (2019a; 2019b)
NH_3_sul_amo	Kg NH_3 kg N^{-1}	17.00%	1.82%	42.50%	EEA (2019a; 2019b)
NH_3_org	Kg NH_3 kg N^{-1}	8.00%	3.04%	20.00%	EEA (2019a; 2019b)

BM: baseline modelling; AM: alternative modelling.

For the present study, it is of interest to compare the uncertainty results between the two methodological approaches and for each of the productive models analysed. Following this idea, the uncertainty in the calculated impacts was estimated from the explicit uncertainty range of the emission factors used in the modelling approaches analysed (BM and AM). Due to lack of information in the literature used for the emission factors, latent uncertainties in NO_3^-, PO_4^{3-} emissions and pesticide fates were not considered for the uncertainty estimation. Consequently, only the explicit uncertainty for N_2O, NH_3 and NO_X emissions was considered (table 7).

Table 8. Results of the Monte Carlo simulations of conventional system, with goblet spur pruning, rainfed, Bobal variety (CRB)

Impact categories	Mean	Variation coefficient	10%	90%
Climate change, default, excl biogenic carbon BM	$1.46 \cdot 10^{-1}$	33%	$8.81 \cdot 10^{-2}$	$2.13 \cdot 10^{-1}$
Climate change, default, excl biogenic carbon AM	$8.22 \cdot 10^{-2}$	22%	$6.27 \cdot 10^{-2}$	$1.07 \cdot 10^{-1}$
Fine Particulate Matter Formation BM	$9.26 \cdot 10^{-4}$	26%	$6.32 \cdot 10^{-4}$	$1.25 \cdot 10^{-3}$
Fine Particulate Matter Formation AM	$6.64 \cdot 10^{-4}$	23%	$4.78 \cdot 10^{-4}$	$8.70 \cdot 10^{-4}$
Photochemical Ozone Formation, Ecosystems BM	$2.66 \cdot 10^{-3}$	52%	$1.05 \cdot 10^{-3}$	$4.57 \cdot 10^{-3}$
Photochemical Ozone Formation, Ecosystems AM	$9.66 \cdot 10^{-4}$	37%	$5.59 \cdot 10^{-4}$	$1.45 \cdot 10^{-3}$
Photochemical Ozone Formation, Human Health BM	$2.65 \cdot 10^{-3}$	53%	$1.04 \cdot 10^{-3}$	$4.57 \cdot 10^{-3}$
Photochemical Ozone Formation, Human Health AM	$9.60 \cdot 10^{-4}$	37%	$5.54 \cdot 10^{-4}$	$1.45 \cdot 10^{-3}$
Stratospheric Ozone Depletion BM	$3.55 \cdot 10^{-6}$	51%	$1.40 \cdot 10^{-6}$	$6.00 \cdot 10^{-6}$
Stratospheric Ozone Depletion AM	$1.18 \cdot 10^{-6}$	56%	$4.61 \cdot 10^{-7}$	$2.10 \cdot 10^{-6}$
Terrestrial Acidification BM	$4.61 \cdot 10^{-3}$	38%	$2.44 \cdot 10^{-3}$	$7.05 \cdot 10^{-3}$
Terrestrial Acidification AM	$3.38 \cdot 10^{-3}$	37%	$1.92 \cdot 10^{-3}$	$5.01 \cdot 10^{-3}$

AM: baseline modelling; BM: alternative modelling.

The variation coefficient was used as a proxy variable to describe the uncertainty in each impact category in which the estimates are susceptible to changes due to changes in NH3, NO_X and N_2O emissions. The variation coefficient is a relative dispersion statistic which allows the variability experienced in several models to be compared. The variation coefficients were obtained by applying Monte Carlo simulations to each production model within the framework of each modelling approach. The contribution to the uncertainty of the emission factors was assessed by means of 5,000 runs of the Monte Carlo simulation using the GaBi v. 9.2.0.58 Analyst Tool.

Table 8 shows the results of the Monte Carlo simulation for the CRB production system, whereas the results for the remaining productive systems are shown in SM-3. From the variation coefficient two patterns can be observed. On the one hand, for CRB, CRT, ORB, ORT, OIB and OIT systems, higher coefficients with BM versus AM are observed, except in SOD. On the other hand, in CIB and CIT production systems, variation coefficients are also higher in BM, although in this case the exceptions are FPMF and TA. In general terms, these results show that the quantified uncertainty is greater when using BM versus AM. This supports the results of table 6, which shows the relevance of using the AM approach to analyse impact categories such as CC, FPMF, POFe, POFh and TA.

4. Discussion

In the context of this study, the Bobal cultivar is found to have a better environmental profile than the Tempranillo, due to the former's higher yield. Specifically, the results permit the organic production of Bobal grapes to be recommended as a feasible alternative to mitigate the environmental damage associated with farming. Nevertheless, in the cases of POFh and POFe, the conventional gs-rainfed systems are a better environmental alternative than the organic gs-rainfed ones. As regards FPMF and TA, not enough evidence was found to support the statement that the conventional gs-rainfed systems generate a greater environmental impact is than the organic gs-rainfed ones. Along these lines, there is a wide margin for improvement; in the short and medium term one proposal could be to replace conventional Bobal crops (approximately 60.5% of the agricultural area of Utiel-Requena) and conventional Tempranillo crops (approximately 9.7% of the agricultural area of Utiel-Requena) for their peers in organic farming. In the transition of Tempranillo crops from conventional to organic, it is recommended to start by changing CIT to OIT because CIT is the heaviest pollutant of all the systems analysed. Another alternative likely to improve the environmental profile of the Utiel-Requena vineyards is that of changing from the Tempranillo cultivar to Bobal. However, this would require greater technical and economic efforts and this cultivar imparts specific characteristics to wine. It is, thus, worth mentioning that this recommendation only considers an environmental approach and it is sensitive to the inclusion of social, quality and/or economic variables in the analysis.

It is important to state that these results may become sensitive to the functional unit; for instance, one considering the profit associated with each productive model. However, due to the scope of this investigation and the uncertainty and volatility of the economic variables of the productive sector being analysed, a kilogram of harvested grapes was considered as the functional unit.

On the other hand, the significant share of on-field fertiliser and pesticide emissions in most of the impact categories makes the modelling approaches a critical point of special interest when applying LCA to agricultural systems. The results of the inferential analysis indicate that, depending on the environmental impact category being analysed, the use of site-specific methodologies guarantees the precision of the analysis. Generic estimation approaches are presented as a robust alternative in the analysis of CC and ME; in this way, the modelling efforts can be minimised, and the results would be consistent with those of more specific methodologies. However, as to the results of FwE, POFe, POFh, SOD, TA, ET and HTnc, the use of the generic modelling approaches shows a significant overestimation when compared to the site-specific ones. When analysing the influence of the modelling approach on each individual emission, there is a greater consistency in the modelling of the NH_3 and NO_3^- emissions in which no significant differences were found, this is not the case when the rest of the emissions are modelled.

The results of this study are consistent with those from Schmidt Rivera et al. (2017) and Peter et al (2016) insofar as there is an overestimation of the environmental impacts associated with on-field fertiliser emissions using generic modelling approaches as compared to those approaches using site-specific information. In line with that found by Goglio et al. (2018), the results also show that, in the absence of specific information, the application of general models for the purposes of estimating fertiliser emissions when analysing CC, a widely analysed impact category, are not invalidated. Mechanistic models for the simulation of water and nitrogen balances in crops, such as STIC (Brisson et al., 1998) or LEACHM (Wagenet and Hutson, 1989), would also be recommended for the purposes of estimating fertiliser emissions, although greater effort is needed to understand the model and to gather the data. Those models have already been successfully applied in other agricultural LCAs (Perrin et al. 2017 and Fenollosa et al., 2014) and take into account irrigation practices, which is also a decisive factor for NO_3^- emissions.

On the other hand, as to toxicity related impacts, an overestimation of ET and HTnc using the modelling approach proposed by Margni et al. (2002) to estimate on-field pesticide emissions has been found in this study and also in Schmidt Rivera et al. (2017); however, in the reference scenario of Peña et al. (2019), no significant differences were found between these two modelling proposals.

The results obtained in this study, together with those from other authors in other regions and for other agricultural products (Goglio et al., 2018; Peña et al., 2019; Schmidt Rivera et al., 2017), highlight the greater variability in the results obtained when modelling pesticide fate than when modelling fertiliser emissions.

The literature of LCA on vineyards in Italy and Spain, shows that CC is the most analysed impact category (Bonamente et al., 2016; Bosco et al., 2011; Chiriaco et al., 2019; Meneses et al., 2016; Mohseni, Borghei & Khanali, 2018; Neto, Diaz & Machado, 2013; Ponstein et al., 2019; Villanueva-Rey et al., 2014). In addition, the IPCC tier 1 methodology (2006a) is the most used to estimate on-field emissions from fertilisers. When comparing CC results of the literature with those obtained in this study with BM, Neto et al. (2013) and Mohseni et al. (2018) show values of 1.82 and 0.51 kg CO_2 eq.·kg of grapes^{-1} respectively, far superior to those of the present study and the rest of the literature. The average results for this impact category in the rest of the reviewed literature is 0.20 kg CO_2 eq. kg of grapes^{-1} versus the average 0.15 kg CO_2 eq.· kg of grapes-1 of the systems analysed in this study, that is, a 33% difference.

Among the reviewed literature, Falcone et al., (2015) analysed several impacts with ReCiPe 2008 (Goedkoop et al., 2009) to compare organic and conventional vineyard per ha. The results are similar in both studies indicating greater environmental impacts of conventional crops compared to organic ones in the CC, FPMF, ME, POFe, POFh, FD, MD, LU, IR and TA categories.

5. Conclusion

The present study assesses wine grapes in the Utiel-Requena PDO, where vineyards account for 87% of the agricultural area of the municipalities. The results show that, regardless of the grape cultivar, organic systems are more environmentally friendly (e.g., on average, the greatest differences are observed for IR, ME and LU, with impacts 1678%, 648% and 171% lower for organic vineyards). In addition, the results for the Bobal cultivar are better than those for Tempranillo thanks to the higher yield. As to the organic management practices, depending on the impact category, the lowest values were those of both irrigated double guyot cane pruning with trellis and rainfed goblet spur pruning systems. These results underline the need to converge to a single indicator in which most of the environmental implications could facilitate the decision-making related to differentiating between the best and worst environmental profiles in production systems.

The results show that, in some cases, the use of modelling approaches that require generic information can make an estimation of fertiliser and pesticide emissions that is as good as those modelling approaches which use site-specific information. It can be concluded that the choice of the methodological approach to be used depends on the impact categories to be analysed, the availability of information and the characteristics of the fertilisers and pesticides that are being applied. In line with other authors, the results also point to the need for a consensus within LCA practitioners on which methodologies to use in order to estimate on-field emissions as they can affect the LCA results considerably. The suggested approaches using site-specific data involve an agreement between the complexity of the data and the minimization of inaccuracies for the purposes of assessing environmental impacts.

Acknowledgments

The authors gratefully acknowledge the Spanish Ministerio de Economía y Competitividad for the financial support under the project CTM2013-47,340-R and the Universitat Politècnica de València for providing the funds for N.K. Sinisterra-Solís's research contract through Subprogram 1 (PAID-01-18).

Appendix A. Supplementary material

Supplementary material to this article can be found online

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