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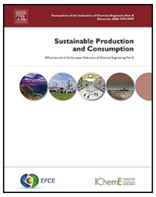

Review article

Towards harmonised and regionalised life cycle assessment of fruits: A review on citrus fruit

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ABSTRACT

The citrus fruit sector is globally relevant. Considering the great contribution of agri-food systems to environmental impacts, assessing and reducing them can make a positive contribution to the environment. Life Cycle Assessment (LCA) is a widespread tool used to quantify the complex environmental interactions of agri-food systems in general and perennial fruit crops in particular. However, methodological aspects need to be harmonised to perform useful and representative LCAs on fruits. The goal of this study is to provide an updated descriptive and critical review of the state of the art of LCA research into citrus fruits. We aim to identify the main methodological decisions, paying special attention to crop cycle modelling and regional representability. Bearing this in mind, we propose recommendations for a harmonised application of LCA on citrus fruits, identifying areas worthy of further research. The main hotspots of the production process are also identified, to understand where improvement efforts should be directed to. To this end, a two-step search was carried out and a final sample of 23 records was obtained. The production of both pesticides and fertilisers together with their on-field emissions are the main hotspots in the reviewed articles. Regarding areas for further research, a lack of studies into the early stages of citrus fruit production is detected. Farm representativeness, both temporal and spatial, is highlighted as a critical issue when assessing regional fruit production. This implies improving life cycle inventories, namely by using site-specific methods to estimate fertiliser and pesticide emissions, developing regionalized datasets of agricultural inputs, and strengthening water inventories. As to the impact assessment, the estimation of both water scarcity and biodiversity impacts is encouraged, together with the use of regionalised impact characterisation methods. Boosting LCA studies on citrus fruits producing countries outside the European Union along with the use of other sustainability tools can support the development of environmental policies. The results of this review can be beneficial for both LCA practitioners and decision-makers, paving the way for a more responsible and sustainable citrus fruit production.

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1. Introduction

Food production will need to increase by around 70 % by 2050 to feed the projected population growth (Bell and Horvath, 2020). This increase in production is of great concern for both the authorities and the consumers themselves (UN, 2019). Ensuring food security requires fundamental changes in the way we produce and consume. In fact, United Nations Sustainable Development Goal number 12, “responsible production and consumption”, seeks to create net gains from economic activities by reducing resource consumption, degradation, and pollution. It thus raises the need to adopt a systemic approach and achieve cooperation between participants in the supply chain, from the producer to the final consumer (UN, 2019). ‘Business as usual’ is no longer an option, hence, achieving a sustainable agriculture is an agreed and essential objective, which requires the application of methods to identify the environmental hotspots of agricultural processes and the implementation of techniques to improve their environmental performance (Nicoló et al., 2015). The agri-food sector is a relevant contributor to environmental impacts via resource depletion, land degradation, air emissions, or waste generation (Beccali et al., 2009) and, ironically, is both a significant contributor to climate change while simultaneously being affected by it (Thornton and Lipper, 2014). Therefore, there is a need to understand the interactions between food security and global environmental change (Ingram, 2012) to propose and adopt solutions towards a sustainable food system.

The global importance of the citrus fruit industry can be highlighted statistically; a total of 143756 thousand tonnes of citrus fruits were produced in 2019, of which approximately 12 % are exported. The main fresh citrus fruit producing region is Asia, accounting for 50 % of global production, with China (25 %) and India (9.3 %) in the lead. South America is relevant as well since is responsible for 19 % of world production, with Brazil (14 %) leading. This country additionally stands out for its high juice production, accounting for 1317 thousand tonnes of frozen concentrated juice in 2020. Within Europe, the Mediterranean region is the main producer, where Spain, Egypt, Turkey and Italy produce 4.2, 3.2, 3.0 and 2.0 % of global citrus fruits, respectively. South Africa is another important actor, producing 2.0 % of citrus fruits globally (FAO, 2021). Given the great importance of citrus fruit production in the agri-food sector, reducing its impacts can contribute positively to the environment. Farmers and managers of agri-food businesses need to understand where these impacts come from and how to deal with them in order to optimise production systems (Martin-Gorriz et al., 2020). Thus, reliable methods are required to identify the impact of the agricultural and horticultural product groups that have the greatest environmental damage potential.

Life Cycle Assessment (LCA) constitutes a recognised and accepted tool that aims to analyse objectively, methodically, systematically, and scientifically the environmental impacts caused by the products from their origins, such as the extraction of the raw materials necessary for their manufacture, until the products are consumed and become waste, through their processing. LCA is increasingly used to evaluate and analyse food environmental issues, but much remains to be done to attain sustainable food security; LCA, in combination with other disciplines, arises as a powerful tool with which to address these issues. It is thus important to expand the assessment of food environmental impacts, also including those of citrus fruits, to more regions and countries, considering the current systems and practices. To this end, methodological aspects should be harmonised as variations in assumptions, methodological choices, inventory data and emission factors used by LCA practitioners could lead to different results, even for similar products, increasing the uncertainty of the impact results (Escobar Lanzuela et al., 2015) and also affecting the comparability of studies (Brandão et al., 2012). Agricultural and bio-based systems are naturally variable due to the variability of climate and other agroecological factors, in addition to uncertainties related to data and process modelling (Brandão et al., 2022). Particularly, when considering the complexities of environmental interactions of agri-food systems in general, and perennial fruit crops in particular, a specific viewpoint on methodological choices and assumptions is required to perform LCAs (Sala et al., 2017). Bessou et al. (2013) reviewed LCA studies on perennial crops, paying particular attention to the farm stage, and made some recommendations for applying LCA to those systems. One year later, Cerutti et al. (2014) reviewed the state-of-the-art practice in LCA on fruit production and described a reference framework for LCA applications in fruit production systems. Among other issues, the authors propose an approach to model the whole life cycle in the orchard, recommending a four-year time interval as a minimum data requirement and advising the inclusion of at least three orchards. Since then, many studies from diverse geographical locations have been published regarding fruit production (e.g. Coppola et al., 2022; Vinyes et al., 2017; Zhu et al., 2018). Generally, fruit LCAs aim to be representative of a specific country or region, which poses methodological challenges related to a huge variability of farming systems and to a lack of data to represent the farm typologies (Pradeleix et al., 2022). In addition, regionalized case studies also require the use of regionalized impact assessments (Morais et al., 2016).

In view of the increasing application of LCA, the updating of methodological issues, and taking into account the complexity and regional particularities found in perennial fruit production systems, an updated review in this area is needed. Specifically, citrus fruits are chosen due to their importance worldwide and since, to the best of the authors' knowledge, there is so far no review study focused on this crop. In sum, this study provides an actualized descriptive and critical review on the state of the art of LCA research applied to citrus fruit, where methodological decisions of the practitioners are presented and discussed thoroughly. Throughout the review process, the authors aim at answering the following key questions:

1. How LCA has been applied in evaluating the environmental impacts of citrus fruit production and its derivatives?
2. In particular, how is the crop cycle modelled and the regional representability addressed?
3. Based on the review, is it possible to obtain recommendations to carry out LCAs into citrus fruits and on perennial fruit crops in general?
4. Which are the life cycle stages with the greatest contribution to each impact category in LCAs into citrus fruit?

With the first two questions we aim to identify trends among the key methodological choices, which will allow us to answer the third question, that is, the proposal of recommendations towards a harmonised application of LCA for regionalised citrus fruits production and, in addition, allow for the identification of areas worthy of further research. The fourth question would benefit the promotion of a more responsible and sustainable citrus fruit production, since hotspots will be detected, and farmers and managers of agri-food businesses will understand where the main changes must be implemented.

2. State-of-the-art of LCA applied to citrus fruit production

2.1. Literature review method

To perform the review, the methodology proposed by Denyer and Tranfield (2009) was used as a guide. The search was carried out considering two main steps (Fig. 1). First, the identification step, where a systematic search in Scopus and Web of Science (WOS) databases was performed to identify the articles to be considered and then a screening and eligibility step, analysing the abstract, introduction and reference sections. In the identification phase, articles, articles-in-press, books, book chapters and proceedings were screened from WOS and Scopus databases to identify scientific publications focusing on the environmental LCA of citrus fruits and derived products. The search strategy was limited to records written in the English language and combined a group of terms related to citrus fruits, namely “citrus”, “orange” and “lemon”, with another group including terms associated with environmental sustainability and LCA; “life cycle assessment”, “LCA”, “sustainability assessment”, “environmental impact assessment” and “environmental sustainability”, focusing their identification either in the title, keywords or abstracts, and with no filter as regards the year of publication. Then, in the screening and eligibility phase, the selected records were revised to ensure their adjustment to the scope of the review.

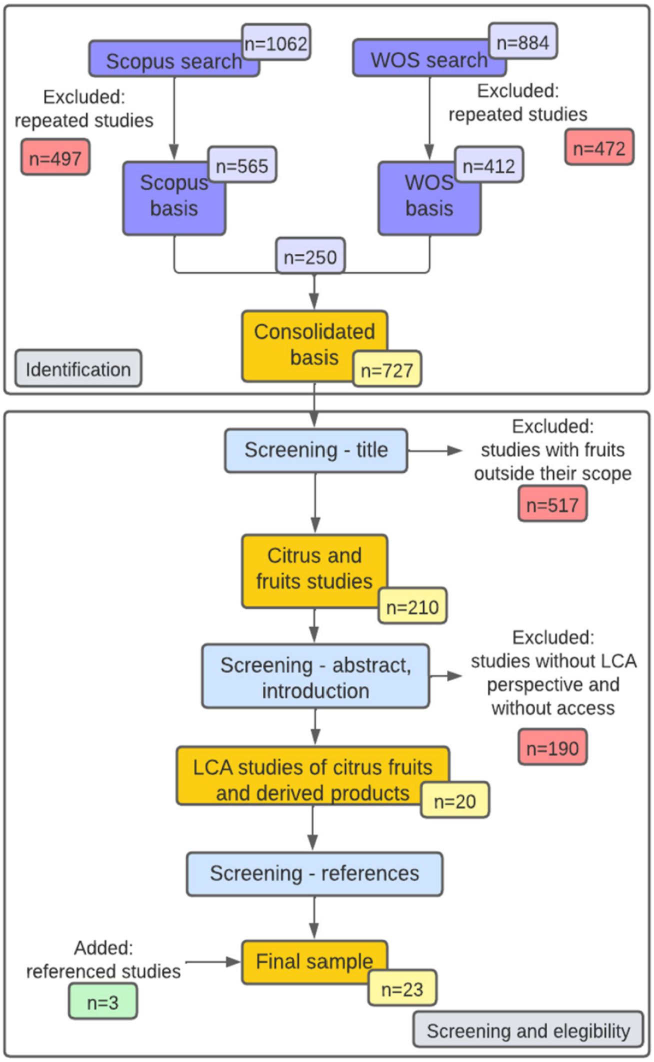

Fig. 1. Review strategy followed to select the LCA studies of citrus fruits based on Denyer and Tranfield (2009).

Given the large number of studies found by applying the aforementioned filters, an initial screening was made to select the articles that explicitly mentioned citrus fruits or fruits in the title ($n = 210$). A second screening was performed on this group of articles, perusing the abstract and introduction, and it was found that 30 % of the articles ($n = 63$) address the use of by-products from citrus fruit processing and were, consequently, excluded. 19 % of the initial sample ($n = 39$) is related to technological aspects linked to fruit sustainability, such as remote sensing or deficit irrigation, but do not constitute environmental impact studies themselves, for which they were also excluded from the review. Finally, 39 % of the 210 studies ($n = 82$) were discarded as they deal with other sustainability issues without an LCA perspective, studying social and economic impacts or consumer preferences. A total of 26 articles (12 %) were obtained. 6 records were excluded as they reviewed or integrated the others. To enrich the search, a third screening was carried out within the references cited in the selected articles and 3 new articles that meet the search criteria, that is, LCA studies on citrus fruits and derived products, were added. After applying the filters, a final sample of 23 research articles remained, which can be divided into two groups: (i) 16 studies on fresh citrus fruits (Table 1), mainly focused on the agricultural stage, although 4 of them also include the postharvest stage; and (ii) 7 studies into citrus-derived products (Table 2), mainly concerning the processing stage, specifically juice and essential oil manufacturing.

3. Results of systematic literature review on LCA of citrus fruits

3.1. General aspects of the LCA studies selected

Data from the reviewed studies have been extracted and combined into several tables. The group of LCA studies on fresh citrus fruits, which is the largest, is also the most heterogeneous (Table 1). It includes research articles ranging from 2009 to the most recent in 2022, which reaffirms the validity of the present research field. As shown in Table 1, most of them focus on western countries, specifically Italy and Spain, which are among the largest citrus fruit producers in the Mediterranean region, as remarked in Section 1. Studies have also been carried out in Brazil, the largest orange producer in South America, and China. Studies in other countries, such as India, Mexico, Iran and Argentina, are more recent. Fig. 2a illustrates the number of studies according to the producing country and the product analysed. To highlight the relevancy of citrus fruit production in those studies, Fig. 2b shows the share of worldwide citrus fruit production of each producing country identified in the review.

Among citrus fruit species, oranges are the most studied, followed by lemons and mandarins, leaving grapefruit aside. This is coherent with the world production ranking, where over 53 % of the production of citrus fruit corresponds to oranges and grapefruits account for under 7 % (FAO, 2021). Only one study into Navel oranges (Nicolo et al., 2017) follows the Product Category Rules (PCRs) for fruits and nuts. The reviewed studies only cover the environmental dimension of sustainability except for four studies (De Luca et al., 2014; Nicolo et al., 2017; Pergola et al., 2013; Ribal et al., 2009), which consider the economic dimension through Life Cycle Costing (LCC). None of the reviewed studies includes the social dimension.

Fewer studies analyse the environmental impacts of citrus derivatives. Beccali et al. (2009, 2010) study the environmental impacts of the production of lemon and orange essential oils and natural and concentrated juice in Italy. Machin Ferrero et al. (2021) and Machin Ferrero et al. (2022)

Table 1
Main methodological choices as refers to the goal and scope of the reviewed LCAs of fresh citrus fruits.

Reference	Producing country	Functional unit	Main goals	System boundaries
Alishah et al. (2019)	Iran	1 kg oranges and 1 ha orange orchard	Assess energy indicators and environmental impacts during the initial 7 years of orange orchards	Cradle to farm gate
Bell and Horvath (2020)	Florida, Mexico, Texas, California, Australia, Chile, South Africa	1 kg oranges	Estimate the impact of cradle-to-market life-cycle seasonal GHG emissions of fresh produce commodities	Cradle to market
Bessou et al. (2016)	Morocco	1 kg fresh fruits (Sidi Aissa clementines)	-Analyse how the partial modelling of the perennial cycle may affect results -Make recommendations on modelling strategy and data needs	Cradle to farm gate (including nursery)
Bonales-Revuelta et al. (2022)	Mexico	1 t of fresh orange	Assess the environmental performance of orange production in Veracruz, Mexico	Cradle to farm gate
Coltro et al. (2009)	Brazil	1000 kg oranges for frozen concentrated orange juice	-Develop a cradle-to-door inventory study of oranges for frozen concentrated juice -Contribute to the development and use of the LCA in Brazil	Cradle to fruit centre entry
De Luca et al. (2014)	Italy	1 ha of planted clementines	Analyse the level of sustainability from an economic and environmental standpoint of different clementine production systems (conventional, integrated, and organic) in the Calabria Region (Italy)	Cradle to farm gate
Lo Giudice et al. (2013)	Italy	“1 p” = production of oranges in an orchard of 10.8 ha in a lifetime of 50 years (13500 t)	-Quantify environmental impacts of integrated production of Tarocco oranges -Assess possible improvements in the production	Cradle to distributor
Martin-Gorriz et al. (2020)	Spain	1 kg citrus fruits (oranges, lemons, and mandarins)	-Quantify environmental impacts and identify key impact factors of irrigated agriculture -Assess farming practices that promote sustainable production	Cradle to farm gate (including nursery)
Nicoló et al. (2015)	Italy and Spain	1 ha	Assess environmental impacts of clementine farming systems (conventional and organic) in Italy and Spain	Cradle to farm gate
Nicolo et al. (2017)	Italy	1 kg packaged oranges	Implement an LCA for the production and packaging of Navel oranges, following the PCRs	Cradle to fruit centre
Pergola et al. (2013)	Italy	1 ha and 1 kg oranges and lemons for fresh consumption	Compare the sustainability of organic and conventional farming methods for lemon and orange through an energy, environmental and production cost analysis	Cradle to farm gate
Ribal et al. (2009)	Spain	1 kg oranges	Assess the eco-efficiency of 24 representative scenarios of citrus production in the Valencian Community	Cradle to farm gate
Ribal et al. (2017)	Spain	1 kg citrus fruits and 1 ha	-Compare the environmental impact of organic and conventional citrus fruits systems in the Valencia region (Spain) Assess the variability within both farming systems	Cradle to farm gate
Ribal et al. (2019)	Spain	1 kg oranges	-Analyse the variability in the carbon footprint of organically and conventionally produced Valencian oranges (Spain) -Determine confidence intervals from small samples and how to calculate the variability of the carbon footprint when the inventory is derived from different sources	Cradle to distributor
Yan et al. (2016)	China	1 ha, 1 kg oranges (among other fruits), 1 g vitamin C, 1 dollar	-Quantify the carbon footprint of China's orange production (among other fruits) to assess the contributions of different farm inputs -Generate information for policymakers so they can identify key options to reduce GHG emissions	Cradle to farm gate
Yang et al. (2020)	China	1 ha and 1t fresh citrus fruits production	-Quantify and locate the environmental cost of citrus fruits production using the LCA method -Test the potential of reducing environmental costs by addressing the problems detected through field demonstrations	Cradle to farm gate

Table 2
Main methodological choices as refers to the goal and scope of the reviewed LCAs of citrus-derived products.

Reference	Producing country	Functional unit	Main objectives	System boundaries
Beccali et al. (2009)	Italy	1 kg of each citrus fruit product (fruits, juices, essential oils)	Estimate environmental impacts of the citrus fruits chain	Cradle to distributor
Beccali et al. (2010)	Italy	1 kg of each orange and lemon-based final product (natural and concentrated juice and essential oils)	Assess environmental impacts of citrus fruits production and transformation processes to identify the most significant issues and suggest options for improvement	Cradle to distributor
Dwivedi et al. (2012)	USA	A Not From Concentrate (NFC) orange 1.893 L juice carton	Assess the global warming impact of not-from-concentrate orange juice produced in the state of Florida	Cradle to consumer
Knudsen et al. (2011)	Brazil - Denmark	1 L of organic orange juice imported to Denmark (for the analysis of organic orange juice) 1 t of oranges leaving the farm gate (for the comparison of orange production processes)	-Identify the environmental hotspots in the production chain of organic orange juice -Compare environmental impacts of organic and conventional orange production	Cradle to distributor
Machin Ferrero et al. (2021)	Argentina	1 t of lemons transported to the factory 1 t of each final product at the factory gate (fresh fruit, essential oil, clarified concentrated juice, cloudy concentrated juice, and dehydrated peel)	-Estimate the WF profile of lemons and lemon-derived products in Tucumán -Identify the parts of the production system that contribute the most to its environmental impact and infer process options that reduce this impact	Cradle to factory gate
Machin Ferrero et al. (2022)	Argentina	1 t of lemons transported to the factory 1 t of each product (Scenario A: essential oil, concentrated juice and dehydrated peel Scenario B: essential oil, concentrated juice, limonene and ethanol)	-Present the environmental profile of lemons and derivatives in Argentina -Analyse the environmental implications of shifting from the conventional production scheme to a biorefinery that includes circular economy strategies	Cradle to the entrance of the factory Cradle to factory gate Cradle to market
Roibás et al. (2018)	Pre-processing: Austria, Holland. Final processing: Malta	A 250 ml bottle of packaged orange juice	Calculate the carbon footprint of ten multi-fruit juices marketed in Malta (including orange juice)	Cradle to market

assess the production and processing of lemons in Argentina to obtain juice and other coproducts (e.g. essential oil and dehydrated peel, among others). Knudsen et al. (2011) assess the manufacturing of organic orange juice in Brazil that is then exported to Denmark. Roibás et al. (2018) calculate the carbon footprint of orange juice marketed in Malta, while Dwivedi et al. (2012) assess the global warming impact of concentrated orange juice produced in Florida (USA).

3.2. Review of the main methodological choices

In this section, by considering the two first research questions defined, the main methodological choices implemented in the reviewed papers are presented (Fig. 3). As commented in Section 1, special emphasis is made on both crop cycle modelling and data representativeness.

3.2.1. Goal and scope

Each of the LCAs reviewed aims to quantify the environmental impact of citrus fruits, with some nuances depending on the case study (Tables 1 and 2). As for the studies on fresh citrus fruits (Table 1), the main goal is to assess the environmental impacts of the crop, although some authors also seek to propose practices to improve sustainability or compare production systems. Some studies, in addition, make a methodological contribution to enrich LCA practice and result analysis. Among them, Bessou et al. (2016) analyse how the partial modelling of the perennial crop cycle through non-holistic data collection may affect LCA results. Ribal et al. (2017) assess the variability of the environmental impacts due to the variability of agricultural management practices by applying a bootstrap technique. Whereas Ribal et al. (2019) go a step further and analyse the variability in the subsequent life cycle stages and how they influence the carbon footprint of oranges.

Taking into account that the main function of agricultural systems is providing food, a mass-based functional unit (e.g. 1 kg or 1 t) is mostly used (Table 1), which reflects the effects of yield on the environmental impacts. Only De Luca et al. (2014) and Nicoló et al. (2015) consider an area-based functional unit (1 ha), which refers to the ability of agricultural systems to provide ecosystem services. Several studies combine mass with area-based analysis. Yan et al. (2016) explore other functional units and use vitamin C content, as a characteristic of food that influences its commercial value, and the dollars earned from the sale of the product. This kind of functional unit, however, is heavily influenced by the economic context (Cerutti et al., 2014).

Temporal boundaries are a relevant aspect for these types of studies since citrus fruits are perennial crops, and thus they present different growth phases. As Ribal et al. (2017) specify, the total life period of an orchard is around 50 years; the first 6–7 years correspond to the low productivity phase and the following 30–35 years to the full production phase. Later, when the yield decreases (senescence phase) the trees are usually uprooted due to economic reasons. Both the dose of applied inputs and the resulting yield vary according to the growth stage.

Most of the studies set ‘cradle to farm gate’ system boundaries (Table 1), where allocation does not make sense because only one product, harvested oranges, is obtained. As regards the studies which include the postharvest stage where oranges are classified, only Ribal et al. (2019) and Machin Ferrero et al. (2021) allocate the environmental loads. The former between the main product (commercial oranges) and the by-products (oranges with unstable or stable defects, used for fodder or sold for juice manufacturing, respectively) performing an economic allocation following PAS 2050–1 guidelines (PAS 2050, 2012) and considering the price of each product when leaving the post-harvest centre. Machin Ferrero et al. (2021) consider a mass allocation between lemons destined for fresh consumption and those that will be further processed, neglecting lemon losses.

As to capital goods, most of the studies do not refer to their inclusion or exclusion and only five studies give a rationale for their exclusion.

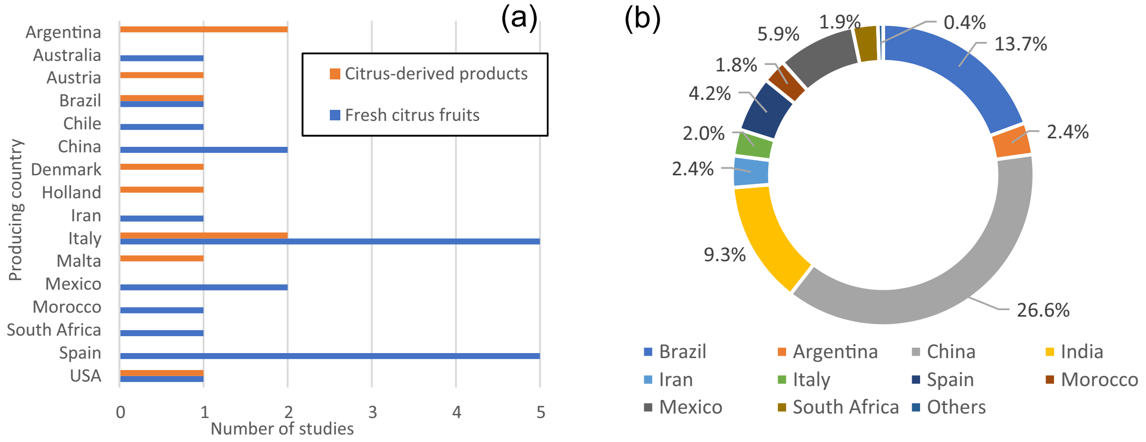

Fig. 2. (a) Number of LCA studies according to the producing country and the product analysed. (b) Percentage of worldwide citrus fruit production corresponding to each producing country identified in the review. 'Others' are countries with negligible contribution (USA, Australia, and Chile).

Namely, Ribal et al. (2009), Roibás et al. (2018), and Knudsen et al. (2011), because their long life means that they have minor impacts on the results. Whereas in Nicoló et al. (2015) and Ribal et al. (2017), the machinery is mostly rented, therefore its impact is also negligible as the use is more intensive than if it was used only on the studied farm.

The main goal of the LCAs of citrus-derived products is to assess the environmental impacts by identifying the main critical points (Table 2), and only Beccali et al. (2010) and Machin Ferrero et al. (2021) suggest improvement options. As far as the functional unit is concerned (Table 2), both volume (for juice) and mass of the final products are used. As to temporal boundaries, almost all the studies that provide this information use data from 1 year. Only Machin Ferrero et al. (2021) use data corresponding to the period 2012 to 2018.

Concerning the life cycle stages included in the system boundaries, a 'cradle to market/distributor' approach is considered in most of the studies, and only Dwivedi et al. (2012) assess the consumption stage (Table 2). A variety of allocation procedures can be found for these products. Roibás et al. (2018) allocate all the annual inputs and outputs to the final product based on their annual mass production. Knudsen et al. (2011) handle the environmental burdens between the frozen concentrated orange juice and the by-products by subtracting the avoided environmental burden of producing barley, a marginal representative for carbohydrate fodder. Beccali et al. (2009) apply mass or economic allocation depending on the process stage and the product, whereas in Dwivedi et al. (2012), only mass allocation is applied. Machin Ferrero et al. (2021, 2022) both perform mass and economic allocations based on the rationale that this decision significantly influences results for the lemon derivatives. The most affected is essential oil production, which is obtained in a much smaller mass proportion than other co-products, but its market value (per tonne) is significantly higher, while the opposite happens for dehydrated peel.

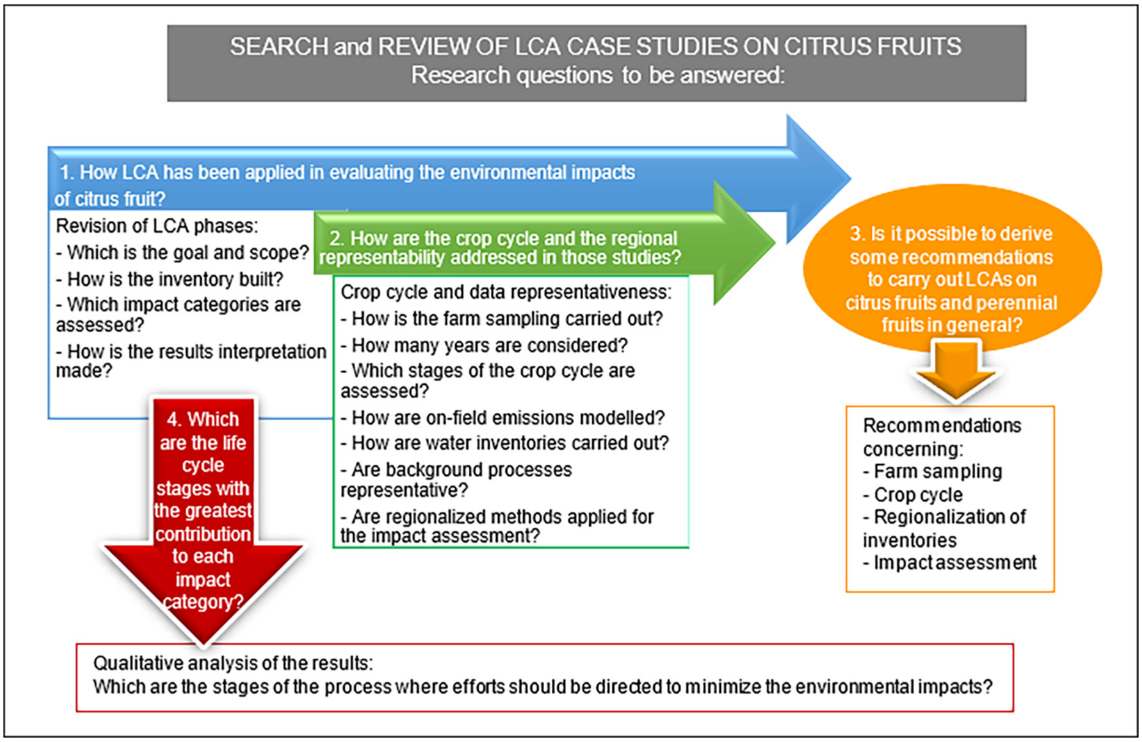

Fig. 3. Main methodological aspects of the LCA revised to answer the research questions.

3.2.2. Inventory analysis

In this section, the data sources used, and their representativeness are discussed. In addition, the methods used for estimating on-field emissions from fertilisers and pesticides and water use, central to the inventories of agri-food products and highly dependent on site-specific characteristics, are reviewed.

3.2.2.1. Data representativeness. One of the main challenges of LCAs is to collect representative data; it is a time-consuming procedure and difficult due to the lack of sources containing detailed or quality-checked datasets, which, in turn, can affect data uncertainty (Beccali et al., 2010). The reviewed studies mostly follow the common LCA practice as to the data sources used. In particular, primary sources are used for the foreground system, that is, central processes under study (mainly farming, post-harvest operations, or juice manufacturing). These are then connected with the background process, corresponding to upstream and downstream life cycle stages (e.g. electricity, fertiliser and pesticide production, transport), which are taken from secondary data sources. Only Bell and Horvath (2020) and Martin-Gorriz et al. (2020) use secondary data for both the foreground and background systems. This can be explained by the fact that these studies aim to assess the average environmental impact of citrus fruit production in a region, namely California (USA) and Murcia (Spain), instead of studying specific farms or factories. In addition, Martin-Gorriz et al. (2020) state that the quality of all input data was evaluated following the ILCD Handbook (Joint research center, 2011) requirements and was classified as “high-quality”.

Primary data are mainly gathered through direct questionnaires and interviews with farmers or other workers, offline surveys, or direct measurements. When assessing the agricultural production in a region, farm representativeness is a key issue and, as Avadí et al. (2016) state, the quality of the data gathered will influence the final accuracy of the impact results. The principal characteristics of the orchards sampled in the reviewed studies are presented in Table 3 as concerns both the number of farms and the number of years sampled. To take into account the interannual variability, Bessou et al. (2016) emphasize that choosing one single year of production can lead to highly uncertain results, especially in the case of alternating yield. However, most studies focus on one farming season corresponding to the full production years. Among them, Ribal et al. (2019) justify their decision on the grounds that not all the citrus fruit varieties present alternating yield and that it can be reduced through practices, such as chemical and manual thinning, a widespread technique among farmers.

Most of the reviewed authors consider sample representativeness by retrieving data from a typical farm in the region under study although they do not justify how the representativeness of the assessed farm is guaranteed. In addition, they do not explicitly state which growth stages are evaluated and, among those who do so, it is mostly the full production stage that is assessed. Three studies (Bessou et al., 2016; De Luca et al., 2014; Pergola et al., 2013) evaluate the complete cycle by extrapolating data from specific years. Alishah et al. (2019) focus specifically on the initial 7 years of cultivation due to the variable application of agricultural inputs during this period, and Machin Ferrero et al. (2021) gather data from 6 harvest seasons. In addition, Bessou et al. (2016) analyse three alternative modelling choices for the perennial crop cycle. The first is chronological modelling throughout the complete crop cycle, collecting data over the first 9 years of the crop, which includes the non-productive phase (1–3 years), and the increasing-yield phase (4–9 years), and using averaged data corresponding to years 7, 8 and 9 as a proxy for the full production years (10–25 years). For the second model, they use 3-year average data from the full production phase. The third modelling approach covers a selection of different single years from the last three years recorded (7, 8 and 9). They conclude that the share of non-productive years in the environmental impacts of perennial crops is considerable and should thus be included. De Luca et al. (2014) gather data from 5 years, which are extrapolated to the whole clementine orchard cycle (40 years), whereas Pergola et al. (2013) extrapolate data from 4 years to the whole orchard period of 50 years. In no case the authors specify to which crop maturity stage the years used for the extrapolation correspond.

The productive cycle of citrus fruits includes a nursery stage and, according to Bessou et al. (2013) and Cerutti et al. (2014), this stage must be accounted for, as well as the transport of the seedlings to the orchards. Despite this, only Martin-Gorriz et al. (2020) and Bessou et al. (2016) analyse this stage, the latter as secondary data, using the Ecoinvent process “seedling production for fruit tree”. Whereas Martin-Gorriz et al. (2020) adapt the same process to the input consumption in the area of study, although they do not provide information on which data of the process is modified. The results obtained when assessing this stage are key to decide whether its inclusion is relevant when assessing citrus fruit production. In this respect, Bessou et al. (2016) state that the share of this non-productive stage to terrestrial ecotoxicity is non-negligible and Martin-Gorriz et al. (2020) highlight nursery as an important stage, although no explicit comment on its contribution to the impact results is made.

The analysed studies on citrus-derived products cover two stages, the farming and the industrial. Except for Knudsen et al. (2011) and Machin Ferrero et al. (2021, 2022), data for the farming stage is gathered from literature, as it is regarded as a background process. For the industrial phase, all the authors use data from one processing plant, except for Dwivedi et al. (2012) and Machin Ferrero et al. (2021, 2022), who work with data from three companies. Beccali et al. (2009) and Beccali et al. (2010) chose a representative company of the region with an annual production close to the regional average for one year. In this regard, Ribal et al. (2019) highlight the “many to few” relationship, with many farmers and few processing companies, representing the current structure of global food supply chains.

3.2.2.2. On-field emissions from fertilisers and soil management. The estimation of on-field emissions from fertilisers is crucial in agricultural LCAs since these have a considerable weight in the impact results and a variety of methods is available, where the most applied do not take into account the specificities of the studied site. In the following paragraphs, the methodologies used are reviewed in order to observe trends, although not all the literature reviewed explicitly states the methods used to calculate these emissions. Almost all the studies estimate nitrous oxide (N_2O) emissions by using the coefficients proposed by the Intergovernmental Panel on Climate Change (IPCC). Ribal et al. (2009) and Alishah et al. (2019) follow Brentrup et al. (2000), which in turn is based on the 1996 IPCC (Houghton, 1997), whereas the remaining studies use the Tier 1 emission factors of IPCC (2006). Machin Ferrero et al. (2021, 2022) use the emission factor proposed by Renouf (2006) (6.7 % of applied N) who study sugarcane production in Queensland, although their study is located in Argentina.

To estimate the ammonia (NH_3) emissions, various methods of different complexity are applied. Alishah et al. (2019) and Bonales-Revuelta et al. (2022) apply the Tier 1 emission factor from the IPCC (2006), whereas Martin-Gorriz et al. (2020) and Ribal et al. (2017) follow the Tier 2 method from EMEP/EEA guidebook (EEA, 2013). Nicoló et al. (2015) and Ribal et al. (2009) follow Brentrup et al. (2000), who estimate these emissions based on parameters such as temperature, infiltration rate and pH. Knudsen et al. (2011) estimate these emissions as 4 % of N-fertiliser input, based on a study into N losses in Brazilian citrus fruits by Cantarella et al. (2003) whereas Yang et al. (2020) assume that 11.1 % of N- fertiliser input is lost as NH_3, based on Ti et al. (2015), who made a nitrogen balance using data corresponding to China. Other authors use emission factors for countries different to those assessed, without giving a clear rationale for this selection. Among them, the Argentinian case studies (Machin Ferrero et al., 2022; Machin Ferrero et al., 2021) refer to Renouf (2006), who assesses emissions from sugarcane production in Queensland (Australia) and estimates them as 14.9 % of N-urea applied.

Table 3
Main characteristics of the orchards assessed in the reviewed LCAs of fresh citrus fruits.

	Data source	Sample representativeness	Number of years sampled	Growth stages assessed
Alishah et al. (2019)	51 questionnaires	Cochran's formula is used to obtain the sample size	7 years	Initial 7 years of cultivation
Bell and Horvath (2020)	Secondary data sources (enterprise budget reports and literature averages)	Not specified	Not specified	Not specified
Bessou et al. (2016)	1 small citrus fruit orchard	The orchard represents recent production technologies	9 years (3 non-productive years and 6 years of increasing yield) (2000–2008)	Whole cycle. 3-year average of the full production phase. The last 3 years recorded.
Bonales-Revuelta et al. (2022)	Questionnaires to six regional orange farmers	Data from three municipalities of Veracruz, Mexico's largest orange producer state	Not specified	Not specified
Coltro et al. (2009)	30 orange farms. Data was gathered through in-depth questionnaires.	A sampling of 19.5 % of the total orange production area of the State of São Paulo	1 season (2002−2003)	Full production
De Luca et al. (2014)	27 farms. Questionnaires and direct interviews with farmers.	Non-probability sampling with reasoned choice and allocation in stratified sampling, where the three main techniques of cultivation are represented	5 growing seasons (2008–2013)	Full production and the whole cycle
Knudsen et al. (2011)	5 organic small-scale farms, 2 organic large-scale farms, and 6 conventional small-scale farms. Questionnaires and direct interviews.	Organic large-scale farms: 40 % of the volume produced (2 out of 5 farms producing organic oranges for juice in the state of São Paulo)	1 growing season (2006–2007)	4- to 20-year-old productive orange plantations
Lo Giudice et al. (2013)	1 citrus fruit orchard	A reference farm, part of an association of citrus fruits producers	Not specified	Full production
Machin Ferrero et al. (2021)	Interviews and surveys with local experts, lemon growers, and governmental institutions to provide a regional representative sample	Not specified	6 harvest campaigns (2012–2018)	Not specified
Machin Ferrero et al. (2022)	Interviews and surveys with local experts, lemon growers and manufacturers and technical reports	Not specified	Not specified	Not specified
Martin-Gorriz et al. (2020)	Secondary data sources (reliable public sources, scientific studies and Ecoinvent database)	Not specified	Not specified	Not specified
Nicoló et al. (2015)	Spain: 12 organic farms and 11 conventional farms Italy: 9 organic farms and 11 conventional farms. Questionnaires and interviews with farmers.	Not specified	1 growing season (2009–2010)	Not specified
Nicolo et al. (2017)	1 farm (8 plots of 1 ha). Direct interviews, measurements, and secondary data sources	Not specified	1 growing season (2013–2014)	Not specified
Pergola et al. (2013)	4 orchard systems 80 face-to-face interviews and secondary data	Representative farm size with the homogeneous characteristics of the cultivation and environment of the region	4 years (2008–2011)	Whole production cycle[a]
Ribal et al. (2009)	24 representative scenarios	A smallholding representative of current farms in the Valencian Community	Not specified	Full production
Ribal et al. (2017)	142 organic and 123 conventional. Surveys to farmers.	The average area reflects the typical smallholding of the region. Outlier detection technique applied to remove non-representative farms	1 growing season (2012−2013)	Full production
Ribal et al. (2019)	21 organic and 21 conventional. Data from surveys.	Trees in full production. Outlier detection technique applied to remove non-representative farms	1 growing season (2012–2013)	Full production
Yan et al. (2016)	7 orange orchards. Field survey.	Authors claim they chose '5 representative sites'	1 season (2012–2013)	Not specified
Yang et al. (2020)	155 orchards. Surveys to farmers.	Typical citrus fruits orchards of Danling County, southwest China	1 season (2017–2018)	Not specified

[a] The authors do not specify how the whole production cycle was modelled from the sampled years.

Beccali et al. (2009) use data from Goebes et al. (2003), an inventory of ammonia emissions from fertiliser application in the USA, although the study is located in Italy.

Nitrate (NO_3^-) leaching is mostly estimated from nitrogen balances and only Alishah et al. (2019) apply the Tier 1 emission factors from the IPCC (2006). Some authors make their own balances, and others use balance results from official publications or literature. In the first group are Bessou et al. (2016) and Beccali et al. (2009), who make their own nitrogen balances based on Brentrup et al. (2000) and Oenema et al. (1998), respectively. Nicoló et al. (2015) and Ribal et al. (2017) use the results of the Nitrogen Balance in the Valencian Region (MARM, 2010; MAAM, 2014). Ribal et al. (2009) assume that 33 % of applied N is leached, based on the study of Ramos et al. (2002) in the Valencian region, whereas Martin-Gorriz et al. (2020) assume 5 % following the study of Martínez-Alcántara et al. (2012) on the Mediterranean coast of Spain and Machin Ferrero et al. (2021, 2022) assume 6.5 % of the N applied, following Renouf (2006). Knudsen et al. (2011) consider a leaching rate of 15 % of applied N based on studies in central Florida, Israel, and Brazil. Yang et al. (2020) use 9.97 % of applied N, based on the study of Zhao et al. (2010) in Central China.

Few studies estimated nitrogen oxides (NO_x) emissions to air. Bessou et al. (2016), Alishah et al. (2019) and Bonales-Revuelta et al. (2022) follow Nemecek and Kägi (2007), who estimate NO_X as 21 % of N_2O emissions. Martin-Gorriz et al. (2020) refer to Sanz-Cobena et al. (2014), who aim to represent the current Spanish N application rates and practices for croplands. Yang et al. (2020) estimate these emissions as 10 % of the N_2O emissions, based on Perrin et al. (2014) and Machin Ferrero et al. (2021, 2022) estimate them as 5.3 % of applied N based on the study for Australia from Renouf (2006).

As regards phosphate (PO_4^{3-}) emissions, most of the studies apply the SALCA-P model (Nemecek and Kägi, 2007). Yang et al. (2020) use a specific model for China whereas Knudsen et al. (2011) and Beccali et al. (2009) apply models from countries different to the assessed ones, the USA and the Netherlands, respectively. Ribal et al. (2017), on the other hand, do not consider phosphate leaching following Brady and Weil (2008), who argue that the leaching of this compound is very low as mineral surfaces tightly adsorb inorganic forms of soluble phosphorus.

Any change in land use or land management practices (e.g. from tillage to no-tillage) affecting soil organic carbon content must be quantified, both as a measure of soil fertility and to mitigate possible Greenhouse Gas (GHG) emissions. Only Knudsen et al. (2011) estimate changes in soil organic carbon by using the Tier 1 methodology of the IPCC (2006). In the remaining studies, the rationale behind the non-estimation of these emissions could be explained by the fact that changes in soil organic carbon mostly occur within the first 20 years of cultivation (IPCC, 2006), although any reference to this is made.

Pruning residues can also influence on-field emissions depending on their management. These are usually ground and incorporated into the soil, burnt, or used as mulching material. Very few of the reviewed studies specify the management of the pruning residues. Particularly, Lo Giudice et al. (2013) and Pergola et al. (2013) propose scenarios where the residues are left on the ground as mulch and scenarios where they are burned, but the emissions related to those practices are not modelled. Mulching contributes to increase soil organic carbon pool, and as stated above, the effects of this practice are accounted for only when it is recently introduced, that is, less than 20 years; in addition, this practice also reduces water and nitrogen losses, and it should be thus considered when estimating N emissions. On the other hand, pruning waste burning releases emissions to air (biogenic CO_2, NO_x, SO_x, particulate matter) that should also be modelled, although biogenic CO_2 is generally not considered in LCAs. Bessou et al. (2016) report that pruning residues were included in the nitrogen balance, although they do not specify how. Ribal et al. (2009) model the emissions from burning following Van Holderbeke et al. (2004) and conclude that, both in integrated and organic production, shredding pruning residues is preferable to burning them.

3.2.2.3. Primary distribution of pesticides. Pesticides are another relevant issue when assessing the environmental impact of agricultural activities. There are controversies regarding permitted products and maximum limits, as well as the health and environmental consequences of their application. The methods used to apply the pesticides and the climate and soil characteristics influence the primary distribution of pesticides. In particular, concerning their primary distribution immediately after application, assumptions are made, underpinned by different literature sources.

An aspect that comes to light is that most of the studies apply fixed distribution percentages, even after the publication of distribution models such as PestLCI (Birkved and Hauschild, 2006), that accounts for site-specific characteristics. Alishah et al. (2019) follow Van den Berg et al. (1999), who stipulates that 30 % of the chemicals are released into the air, and the rest is transferred to the soil. Ribal et al. (2017) refer to Berthoud et al. (2011), who assume that 0.5 % of the applied doses go to surface water; they then estimate the fraction going to air from the vapour pressure of the pesticide, and finally consider that the remaining fraction goes to soil (with a maximum of 85 %). Machin Ferrero et al. (2021, 2022) assume that the pesticide runoff is 1.5 % of the active ingredient applied, following Renouf (2006). By following Nemecek and Kägi (2007), Bessou et al. (2016) and Bonales-Revuelta et al. (2022) assume that the soil is the final reception compartment of all pesticides. Nicolo et al. (2017) chose to model pesticide distribution following the assumptions of Margni et al. (2002), whereas Nicoló et al. (2015) apply the methodology developed by Hauschild (2000), who contemplates various dispersion routes for the applied pesticides with redistribution factors for the different routes.

3.2.2.4. Water inventory. Accounting for water flows is a prerequisite for the assessment of the impacts associated with water use and again there is a variety of available tools with which to carry out the inventory of water flows in agricultural LCAs (Payen et al., 2018). In this section, the methods used to estimate water withdrawal for citrus fruit irrigation at the farm level and at the processing plant are reviewed. Most of the studies elicit data from direct sources, such as questionnaires or interviews, with some exceptions which are detailed next. Bessou et al. (2016), Martin-Gorriz et al. (2020) and Beccali et al. (2009, 2010) estimate the amount of irrigation water using models based on crop water requirements considering agro-meteorological information. Ribal et al. (2009) take the recommended water dose, for both furrow and drip irrigation systems, for integrated citrus fruit production in the Valencian region according to the official regulations (DOGV, 2001). For organic production, these authors assume an 8 % reduction in the water dose since, in that case, the content of organic matter in soil is greater, which increases its water retention capacity. Nicoló et al. (2015) and Ribal et al. (2017) exclude water from their inventories due to a lack of reliable data and Bonales-Revuelta et al. (2022) assume that irrigation water is negligible as, according to farmers, precipitation provides all their water requirements. As to citrus fruit processing, Knudsen et al. (2011) and Machin Ferrero et al. (2021, 2022) obtain the information from questionnaires and interviews. Nicolo et al. (2017) measure the amount of water used in the postharvest central with a water flow meter. Beccali et al. (2009) and Beccali et al. (2010) estimate the water used at the processing plant considering the operation time and power of the equipment. Roibás et al. (2018) use data from the corporation responsible for the complete drinking and wastewater cycle in the Maltese Islands, where the juices are processed.

3.2.2.5. Background data. As commented in 3.2.2.1., secondary data is used for background processes. Within the secondary sources used in the reviewed studies, estimations and approximations, reliable public sources, LCA databases, scientific studies and other databases can be

Table 4
Impact categories employed in the reviewed LCAs of fresh citrus fruits. The life cycle stages contributing the most to each impact category are highlighted.

Reference	Impact Assessment method	Midpoint Impact Categories												
		GWP	AP	EuP	MDP	FDP	ODP	POP	TEP	FWAEP	MAEP	HTPc	HTPnc	WU
Alishah et al. (2019)	CML2001	FE-FP	FP-FE-PP	FE	FP-PP	FP-DC	PP-FP-DC	FP-PP-FE-DC	FE	FE	FE-FP	FP-PE		
Bell and Horvath (2020)	Literature	T-PS	–	–	–	–	–	–	–	–	–	–	–	–
Bessou et al. (2016)	ReCiPe 2008	FE-I	FE-I	FE-I	–	–	–	–	PE	I-PE	–	I	I	I
Bonales-Revuelta et al. (2022)	CML2001	FE Co: DP - DC	FE-FP-FO	FE-FP-FO	PP-FP		FP-DC-PP-FO	FP-DC	O: FE Co: PE-DC	Co: PE O: FP-FE-LPS	FP	FP Co: PP		FP Co: PP
De Luca et al. (2014)	ReCiPe Eco-Indicator 99	FE-FP	–	–	–	–	–	–	–	–	–	–	–	I
Lo Giudice et al. (2013)	IMPACT 2002+	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	–
Martin-Gorriz et al. (2020)	CML2001	EP-FO	I-FP-FO	FO-I-FP	PP-FP	I-FO	–	–	–	–	–	–	–	I
Nicoló et al. (2015)	CML2001 USEtox	Co: FP O: FP-FO-MP	Co: FP O: FO	Co: FP-FO O: FO	FP-PP	Co: FP O: FP-FO-MP	Co: FP O: FP-PP	Co: FP O: FP-FO-MP	–	Co: FO O: FP-FO	–	FP	Co: FP O: FP-MP	–
Nicolo et al. (2017)	EPD 2015	FO-FE	FO-FE	FE-FO	–	FO-FE	FE-FO	FE-FO	–	–	–	–	–	–
Pergola et al. (2013)	CML2001	Co: FP O: FE-I-FO	Co: FE-FP	Negative numbers due to MA	Co: FE-FO O: FE-FO-I	–	–	Co: FE-FO O: FE-FO-I	–	–	–	–	–	–
Ribal et al. (2009)	CML2001	PB[a]-FP-FE	FP-FE	Eco: FE In: FP	FP	–	FP-PP-MU	PB[a] In: FP Eco: FO	–	–	–	–	–	I
Ribal et al. (2017)	CML2001 USEtox	Co: FP O: MA-MU	Co: FE O: MA	Co: FE O: MA	Co: FP O: MU-PP	Co: FP O: MU-PP	Co: FP O: PP	Co: FP O: MU	–	Co: PE O: MU-PP	–	Co: FP O: MU-PE	Co: FP O: MU-PE	–
Ribal et al. (2019)	CML2001	[b]	–	–	–	–	–	–	–	–	–	–	–	–
Yan et al. (2016)	Literature and IPCC 2007	FE-PE	–	–	–	–	–	–	–	–	–	–	–	–
Yang et al. (2020)	Literature	FP-FE	FE	FE	–	–	–	–	–	–	–	–	–	–

Impact categories: GWP = Global Warming Potential; AP = Acidification Potential; EuP = Eutrophication Potential; MDP = Mineral Depletion Potential; FDP = Fossil Depletion Potential; ODP = Ozone layer Depletion Potential; POP = Photochemical Oxidation Potential; TEP = Terrestrial Ecotoxicity Potential; FWAEP = Freshwater Aquatic Ecotoxicity Potential; MAEP = Marine Aquatic Ecotoxicity Potential; HTPc = Human Toxicity Potential, cancer; HTPnc = Human Toxicity Potential, no cancer; WU=Water Use. Process stages: T = Transport; FE = Fertiliser Emissions; FP=Fertiliser Production; MU = Machinery Use; PP=Pesticide Production; PE = Pesticide Emissions; EP = Energy Production; I = Irrigation; FO = Field Operations; MA = Manure Application; PB = Pruning Burning; DP=Diesel Production; DC=Diesel consumption; LPS = Land Preparation Stage; MP = Machinery production; PS=Production Stage; EU = Energy Use. Production system: Co = Conventional: O = Organic; In = Integrated; Eco = Ecological.

[a] The authors do not consider CO_2 fixation by the trees, but they do consider the CO_2 released into the atmosphere from the burning of pruning waste since the latter occurs over a shorter time than the former.

[b] The study does not explicitly state which stages contribute the most to the impact category.

Table 5

Impact categories assessed in the reviewed LCAs of citrus-derived products. The life cycle stages contributing the most to each impact category are highlighted.

Reference	Impact Assessment method	Midpoint Impact Categories											
		GWP	AP	EuP	MDP	FDP	POP	TEP	FWAEP	MAEP	HTPc	HTPnc	WU
Beccali et al. (2009)[a]	CML2001	FU1: PP-FP FU2: PP-FP-T FU3 and FU4: PP-FP-EP FU5: T-PP-FP-PE-FE FU6: EP-T	FU1: PP-FP-PE-FE FU2, FU3, FU5 and FU6: T FU4: PP-FP-PE-FE-EP	All FUs: PP-FP-PE-FE	–	–	FU1 and FU4: EP-PP-FP-PE-FE-T FU2, FU3, FU5 and FU6: T	–	–	–	–	–	–
Beccali et al. (2010)	CML2001	[b]	[b]	[b]	–	–	[b]	–	–	–	–	–	[b]
Dwivedi et al. (2012)	GHG emissions from literature Global warming potentials from the TRACI database	EP-FE	–	–	–	–	–	–	–	–	–	–	–
Knudsen et al. (2011)	EDIP97	T-PS	T	PS	–	T-PP-FP	–	–	–	–	–	–	–
Machin Ferrero et al. (2021)[a]	AWARE ReCiPe 2016 IMPACT World +1.25 (Roy et al., 2012, 2014) USEtox 2.0	–	FU4 and FU7: PP-FE FU6 and FU8: EP	All FUs: PP-FE	–	–	–	–	FU4, FU6: and FU8: PP-PE FU7: PM- PP-PE		All FUs: PE	All FUs: PE	All FUs: I
Machin Ferrero et al. (2022)[a]	ILCD	FU7: FE FU4: PS-EU	FU7: FE FU4: PS	FU7: PP-FE FU4: PS	FU7: PP FU4: PS	FU7: PP FU4: PS	FU7: DC-FE FU4: PS	FU7: DC-FE FU4: PS	FU7: PP FU4: PS-EU	FU7: FP FU4: PS	FU7: PP FU4: PS-EU		FU7: I FU4: PS
Roibás et al. (2018)	CF from literature	EU	–	–	–	–	–	–	–	–	–	–	–

Impact categories: GWP = Global Warming Potential; AP = Acidification Potential; EuP = Eutrophication Potential; MDP = Mineral Depletion Potential; FDP = Fossil Depletion Potential; POP = Photochemical Oxidation Potential; TEP = Terrestrial Ecotoxicity Potential; FWAEP = Freshwater Aquatic Ecotoxicity Potential; MAEP = Marine Aquatic Ecotoxicity Potential; HTPc = Human Toxicity Potential, cancer; HTPnc = Human Toxicity Potential, no cancer; WU=Water Use. Process stages: T = Transport; FE = Fertiliser Emissions; FP=Fertiliser Production; PP=Pesticide Production; PE = Pesticide Emissions; EP = Energy Production; I = Irrigation; DC=Diesel consumption; PS=Production Stage; PM = Packaging Manufacturing; EU = Energy Use. Products: FU1 = Orange essential oil; FU2 = Orange natural juice; FU3 = Orange concentrated juice; FU4 = Lemon essential oil; FU5 = Lemon natural juice; FU6 = Lemon concentrated juice; FU7 = Fresh lemons; FU8 = Lemon dehydrated peel.

[a] The stages with the highest impact depend strictly on the functional unit considered.

[b] The study does not explicitly state which stages contribute the most to the impact category.

highlighted. Ecoinvent (Ecoinvent, 2022) and GaBi professional (Sphera, 2021) are generic databases widely used, although other specific databases for food systems have been also identified, such as Agrifootprint (Blonk Consultants, 2019) and LCA Food (Nielsen et al., 2003).

An aspect that cannot be overlooked is the observation made by some authors on the completeness of the databases used to model background processes. Knudsen et al. (2011) highlight the absence of country-specific datasets for the production of mineral fertilisers and truck transport. Nicoló et al. (2015) and Ribal et al. (2017) highlight the lack of information corresponding to the production of some active principles of pesticides and inorganic fertilisers in the inventory databases. As to the emissions from the production of organic fertilisers, those from manure, when used, are disregarded in most of the reviewed LCAs, probably because it is considered as waste from livestock activity. However, manure cannot be always considered as waste if it has economic value (Montemayor et al., 2022). Bonales-Revuelta et al. (2022) stand out the lack of inventory datasets for some organic fertilisers and Lo Giudice et al. (2013) state that the machinery used specifically for citrus fruit processing is not present in the chosen database.

3.2.3. Impact assessment methods and impact categories

The impact assessment methods and impact categories evaluated in the reviewed studies are summarised in Tables 4 and 5. All the studies use midpoint approaches, except De Luca et al. (2014), who also use Eco-Indicator 99 as an endpoint approach. The most widely used impact assessment methods are CML2001 (Guinée and Lindeijer, 2002) and ReCiPe (Goedkoop et al., 2013; Huijbregts et al., 2016). Climate change is the most commonly studied impact category, followed by eutrophication and acidification, categories which are closely related to agricultural practices according to the literature. Notwithstanding, any of the reviewed studies applies regionalized methods.

3.2.4. Interpretation of results

3.2.4.1. Methods for comparative analysis. Some of the studies reviewed compare agricultural practices, mainly organic versus conventional, although, in many cases, impact scores are directly compared without applying any consistent method to support the comparison. In particular, Pergola et al. (2013) and De Luca et al. (2014) make a direct numerical comparison of the impact values obtained for conventional and organic citrus fruits using mass and area as functional units, together with Machin Ferrero et al. (2022) who make numerical comparisons between the assessed scenarios (base scenario vs. biorefinery-based schemes, which implement circular economy strategies). Instead, statistical methods are applied in other studies, such as Bonales-Revuelta et al. (2022), who perform a Principal Component Analysis (PCA) to compare the variation between the environmental impacts of every stage of organic and conventional systems. Analysis of variance (ANOVA) is the statistical method applied by Knudsen et al. (2011) and Yang et al. (2020) to compare farm types, where significant differences between groups are considered at $p < 0.05$. Ribal et al. (2017) and Ribal et al. (2019) compare the organic and conventional production of oranges focusing on the farming stage and the whole supply chain, respectively. In both studies, confidence intervals are obtained for the impact results, as they aim to assess the variability in each system, and statistical methods are then applied to compare organic and conventional production. Details on the approaches followed by Ribal et al. (2017) and Ribal et al. (2019) to assess the variability and obtain the confidence intervals are commented on in section 3.2.4.2.

3.2.4.2. Uncertainty and variability assessment. Uncertainty is defined as incomplete or imprecise knowledge, which can be due to troubles in data collection, a lack of detailed information sources, or low data quality, mostly due to the lack of knowledge concerning the actual value of the quantity (Huijbregts, 1998). In the reviewed studies, when quantified, two main methods are used to treat the uncertainty, namely sensitivity analysis and the propagation of uncertainty through Monte Carlo simulation. Sensitivity analysis provides a quantitative means of determining to what extent the results vary when an input changes (Wei et al., 2015). This is done partially, that is, a parameter, assumption or model is changed, whereas the rest is kept constant. Knudsen et al. (2011) perform a sensitivity analysis to elucidate how different assumptions related to the farming stage and processing stages (e.g. manure N content, location of the juice reconstitution plant), and also to modelling choices (e.g. allocation procedures, time perspectives for the modelling of soil organic carbon) affect the results. Bessou et al. (2016) study the sensitivity of the results to modelling choices related to the growth stage of the crop (see Table 3 and 3.2.2.1). Specifically, those authors perform a sensitivity analysis considering different allocation methods, uncertainty related to secondary data sources, and initial assumptions on cultivation, transport, and waste management.

Monte Carlo simulation quantifies the influence of the uncertainty of different input data on the results of the LCA using probability distributions to generate probabilistic results and shows whether both the quality of the collected data and the uncertainty of the results are appropriate (Kroese et al., 2014). It is one of the most applied methods to analyse the uncertainty caused by parameters, although only two of the reviewed studies quantify the uncertainty of the results obtained. In particular, Bell and Horvath (2020) use Monte Carlo simulation to evaluate the uncertainties arising from the variability due to crop yields and life cycle GHG emissions per hectare of the scenarios contemplated. Martin-Gorriz et al. (2020) also use this method to explore the robustness of LCA results for the baseline scenario, as well as to evaluate the change in the results for the impact mitigation strategies proposed. On the other hand, variability can be defined as intrinsic differences over space and time or within a group. Though it cannot be reduced, it can be represented more precisely by gathering more information about the assessed group (Ribal et al., 2017). In general, agricultural systems show a high variability due not only to local factors, such as climate, water availability and quality, or soil type but also to farmers' decisions as regards the agricultural practices to be performed (Bosco et al., 2013). Among the reviewed studies, Ribal et al. (2017) use a bootstrap technique to assess the farming variability. The bootstrap is a resampling technique, in which several Monte Carlo samples of size "n" with replacement are taken from the primary observations. In this way, the authors estimate the distribution of the different impact categories obtained from a group of conventional and organic farms and build confidence intervals to compare the impact values of each type of farm. Ribal et al. (2019) also use a bootstrap technique together with Monte Carlo analysis to assess the variability of the carbon footprint along the supply chain of Navel oranges for both conventional and organic production. They assess the variability of the farming, postharvest and transport to the distribution centre separately and then they calculate the empirical distribution of the total Carbon Footprint (CF) by adding up piecewise each iteration of the three bootstrap procedures of these subsystems.

3.3. Most impacting stages per category

As regards research question number 4, the life cycle stages with the highest score in each impact category are shown in Tables 4 and 5.

In the case of the studies into fresh citrus fruits, fertilisers production and their subsequent emissions are the stages that dominate most of the reviewed categories (GWP, AP, EuP, POP, FWAEP, MAEP). It is therefore recommended to pay special attention to the type and dose of fertilisers applied, especially nitrogenous compounds. The production of pesticides and fertilisers are important stages in the MDP category, given that these activities involve the extraction of mineral resources. The importance of the activities linked to field operations stands out in the FDP and HT categories, as they involve the consumption of fossil resources

Table 6
Updated methods to improve the regional representativeness of inventories and impact assessment in LCAs of fruits.

LCI Input	Modelling	Midpoint impact method	Impact indicator
N-emissions	Tier 3 methods: Daisy (Hansen, 2002) Animo (Rijtema and Kroes, 1991) LEACHN (Wagenet and Hutson, 1989)	ImpactWorld+ (Bulle et al., 2019)	Climate change (kg CO_2 eq) Freshwater acidification (kg SO_2 eq) Marine eutrophication (kg N N-lim eq) Terrestrial acidification (kg SO_2 eq) Photochemical oxidant formation (kg NMVOCeq) Ozone layer depletion (kg CFC-11 eq)
P- emissions	Tier 3 methods: DNDC (University of New Hampshire, 2007) LEACHN (Wagenet and Hutson, 1989) Indigo v3.0 (Bockstaller et al. 2020, submitted)	ImpactWorld+ (Bulle et al., 2019)	Freshwater Eutrophication (kg PO_4 P-lim eq)
Primary distribution of pesticides	PestLCI Consensus model v1.0 (Fantke et al., 2017) Allen et al. (1998)	USEtox (Fantke, 2017) + DynamiCrop (DynamicCROP, 2021)	Freshwater ecotoxicity (CTUe) Human health toxicity (CTUh)
Water consumed in irrigation	CropWat model (Smith, 1992) AquaCrop (FAO, 2016b) (not for citrus fruits) Meteorological databases	AWARE (Boulay et al., 2018)	Water scarcity (m3 world-eq)
Land occupation Land transformation	Chaudhary and Brooks (2018)	Chaudhary and Brooks (2018)	Potential species loss Global: percent disappeared fraction of species Regional: regional species loss

and the exposure to the emissions from the combustion of these resources. As concerns water use (WU), irrigation stands out as the dominant stage. It has to be borne in mind that most of the authors do consider water as an input but do not assess the impact in terms of water scarcity. Only De Luca et al. (2014), Bessou et al. (2016) and Machin Ferrero et al. (2021) assess this impact; the former two by using ReCiPe 2008 (Goedkoop et al., 2013), which is based on the Water Scarcity Index of Pfister et al. (2009), and the latter by applying AWARE (Boulay et al., 2018). As to citrus-derived products, the impact of the farming stage is greater than that of the processing stage, due to the above-mentioned causes. The transportation, either of the agricultural inputs or the final product, and energy production stages are relevant too.

4. Methodological recommendations to harmonize citrus fruit LCAs and increase representativeness

In this section, and to answer research question number 3, proposals regarding methodological issues are made to harmonize the application of LCA to citrus fruits, seeking also to enhance the regionalization of inventories and assessment methods. This is a requirement to guarantee that the LCA framework copes with farm characteristics in the assessed country, which have consequences on the final results (Morais et al., 2016). These proposals, which can also be applied to fruit production from perennials in general, have been made taking into account the PCRs for fresh fruits and juices (EPD, 2019a, 2019b) and other literature sources. Recommendations regarding the improvement of regional representativeness of inventories and impact assessment are summarised in Table 6. It must be noted that, due to its complexities, proposals mostly focus on the farming stage.

4.1. Goal and scope: framing the assessment

Goal definition is pivotal in LCAs because it determines and guides the choices to be made in the subsequent phases of the study (e.g. functional units, system boundaries, data sources, sample representativeness, and impact categories, among others). The choice of the functional unit will obviously depend on the goal of the study. As pointed out in Section 3.2.1, a mass-based unit reflects the main function of agricultural systems and is the functional unit recommended in Environmental Product Declarations (EPDs) according to the PCRs for fresh fruits and juices (EPD, 2019a, 2019b). An area-based unit can

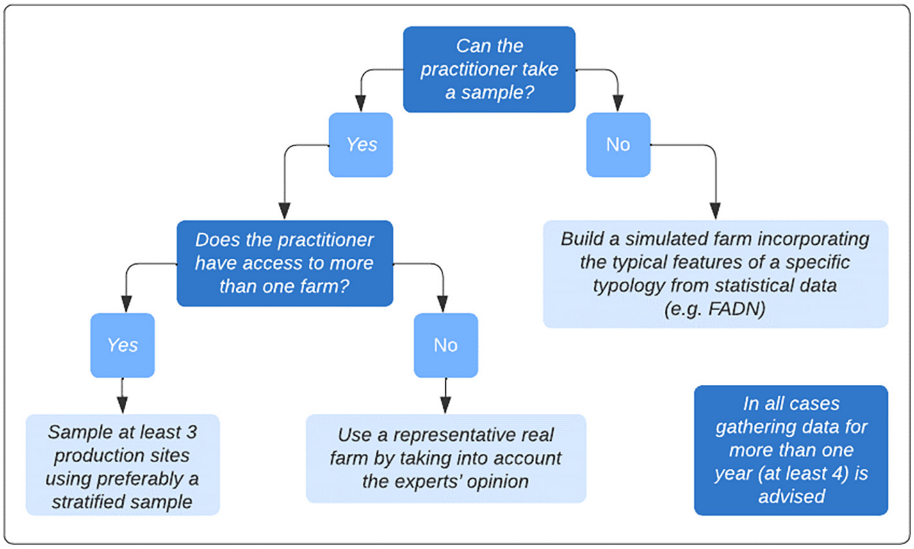

Fig. 4. Recommendations to improve the temporal and geographical representativeness of LCAs of fruits.

hide the influence of agricultural practices on yield (e.g. organic production, intensification, etc.) or the changes in yield in line with the growth stage of the tree. An economic value-based functional unit can be suggested when comparing different commercial fruit categories (e.g. conventional and organic) as it considers the net income gained by fruit growers or product quality (Van Der Werf and Salou, 2015; Yan et al., 2016). The use of a combination of functional units, for example, mass-based and area-based, to compare agronomic systems, avoids the overvaluation of resource use efficiency and the delocalisation of environmental impacts.

System boundaries (also temporal ones) are critical. This review has detected that most of the studies neglect the nursery stage and unproductive years (see 3.2.2.1). According to the PCRs, the emissions and resource consumption of the nursery and the unproductive years must be spread out over the productive years considering the yearly yields and the entire lifetime of the plantation. PCRs claim that if this process is under the direct control of the organization assessed, primary data should be used. In addition, Perrin et al. (2014) state that the nursery stage should be included unless it can be clearly demonstrated that its contribution to the impacts is negligible. In this regard, the two studies that assess this stage cannot be considered conclusive (see 3.2.2.1), as the inventories used are not transparent enough. Hence, LCA studies on the nursery stage from primary data that thoroughly describe the inventory are urged. Following Bessou et al. (2013), a modular assessment, where each stage is modelled independently, is recommended because, given the duration of perennial cropping systems, it is not easy and sometimes even impossible to gather data for the whole cropping cycle. As to capital goods, the EPDs for fresh fruits (EPD, 2019a) state that the technical system shall not include the manufacturing of production equipment, buildings and other capital goods, which is in line with the reviewed studies.

Analyses of the life cycle stages after farming (from farm gate to grave) are often omitted. Although the life cycle stages to be included in the system boundaries depend on the goal defined, the emergence of policies such as the “farm to fork strategy” (CEC, 2020), which seeks to reduce the global footprint of the food system, must be kept in mind. Hence, to support the transition to sustainable food systems more research is required to accurately determine the impact contribution of the post-farm stages, mainly as concerns the case of packaging and transport (Pernollet et al., 2017), as well as the end-of-life, which is hardly included in the literature reviewed. In this regard, PCRs define the attributional processes that are classified as ‘downstream’ and the data requirements related to them.

LCA practitioners have several options when carrying out allocation and must choose the method that best suits the goal of the study, while also considering data availability. The PCRs for fresh fruits (EPD, 2019a) state that fruits and nuts for human consumption, even though they may be of potentially different grades, are considered equivalent in terms of the service they deliver, and allocation between them is thus not needed. In addition, where substandard or waste fruits or nuts are used as animal feed displacement of other feedstock must not be considered. When needed, the optimal choice would be to divide the main process into subprocesses, otherwise, as the PCRs suggest, allocation methods should reflect the physical relationships between products by using mass allocation. When these relationships cannot be determined, economic allocation can be used, as recommended also for the farming stage in the LCAs of juices (EPD, 2019b) and when the co-products do not have similar characteristics and/or functionality (PAS 2050, 2012).

4.2. Inventory analysis and data representativeness

Representativeness involves both temporal and geographical (regional) variability. The first can be captured by gathering data corresponding to different years. Cerutti et al. (2014) recommend collecting field data in an even number of years (at least 4), whereas the PCRs for fruits and nuts do not specify the number of years to be used. Whenever possible, gathering data for more than one year is advised, because even if yield variations are not detected, agricultural practices can change due to external factors (e.g. climate conditions or input prices). As to the fruit processing phase, one year could be enough because, unlike in the farming stage, no variations in the juice manufacturing process are usually observed from one year to another, since the conditions are more closely controlled, making the process more repeatable.

When geographical representativeness comes into play, the selection of a representative sample is required, taking into account all types of farms in the region under study, which can entail complexity (Avadí et al., 2016). To this end, different approaches can be applied, as summarised in Fig. 4, mainly the sampling of real farms (preferably a stratified sample to better represent the types of farms), the selection of a representative real farm, giving a rationale for the selection such as by taking into account the experts' opinion, or the building of a simulated farm incorporating the typical features of a specific typology. In turn, farm typologies can be defined by using statistical methodologies (e.g. clustering or principal component analysis). In practice, the farmers' willingness to share data about their practices can be a limitation (Avadí et al., 2016); thus, selecting or modelling representative farms can be a useful approach when a proper sampling cannot be carried out.

The modelling of on-field emissions is also crucial to obtaining representative inventory data and entails difficulties as these emissions are closely related to site-specific soil and climate conditions. Along these lines, the PCRs for fresh fruits (EPD, 2019a) recommend the use of site or region-specific data, such as Tier 2 and Tier 3 models; otherwise, specific Tier 1 coefficients are proposed. As concerns Tier 1 emission factors, revised coefficients with respect to soil and climate conditions are starting to be developed; IPCC (2019), a refinement of IPCC (2006), disaggregates some of the emission factors by climate region (wet and dry climates) for direct and indirect N_2O emissions. Andrade et al. (2021) compared different Tier 2 and Tier 3 models and recommend Tier 3 models for estimating N_2O and NO_3^- emissions and Tier 2 models for NH_3 volatilisation. In general, Tier 2 models, such as SALCA (Nemecek et al., 2019), are an excellent alternative for reducing complexity and improving precision, although Tier 3 mechanistic models, such as Daisy (Hansen, 2002), Animo (Rijtema and Kroes, 1991) and LEACHN (Wagenet and Hutson, 1989) allow greater certainty in the estimations. These models are also useful to improve the modelling of the emissions from organic fertilisers (Montemayor et al., 2022). It must be noted that mechanistic models require a high amount of input data (e.g. soil composition, application practices, precipitation, etc.) and, due to their complexity, understanding and adapting these models to the region of study requires more effort.

Farmers are encouraged to measure nutrient contents on their soils to obtain more adjusted parameters when using the models. Nevertheless, to help practitioners, there are databases that collect relevant information on soil composition, such as LUCAS (Orgiazzi et al., 2018), which presents information for 23 member states of the European Union, or SIGRAS (INIA-GRAS, 2012), which gives information of Uruguayan soils. Concerning phosphate emissions, Tier 3 models, such as DNDC (University of New Hampshire, 2007), LEACHN (Wagenet and Hutson, 1989) and Indigo v3.0 (Bockstaller et al. 2020, submitted) can be used. To account for changes in soil organic carbon stocks, recent publications (Bessou et al., 2020; Joensuu et al., 2021) test the effect of models of differing complexity on other crops and also make interesting recommendations. As to emissions from pruning leftovers, modelling depends on the management practice. EMEP/EEA (EEA, 2019) gives Tier 1 emission factors for those emissions from the burning of agricultural residues not associated with biogenic carbon. When this residue is incorporated into the agricultural soil, the above-mentioned mechanistic models allow the N mineralization to be accounted for.

The PestLCI Consensus model v1.0 (Fantke et al., 2017), is recommended to estimate the primary distribution of pesticides immediately after their application, as it considers different crop types, plant growth stages, drift deposition curves, and pesticide application methods. Another important aspect related to pesticides to be considered, especially in agri-food LCAs although it has been omitted in the reviewed studies, is the intake fraction, that is, the portion of emitted pesticide that effectively enters the human population, commonly through inhalation and ingestion. In the case of citrus fruit processing (including postharvest treatment), a processing factor must also be considered (Juraske and Sanjuán, 2011), which accounts for the reduction in pesticide residues caused by the processing steps. To this end, the DynamiCROP model is recommended; this is a dynamic plant uptake model that quantifies human exposure and the related health impacts caused by the application of pesticides to food crops and the subsequent ingestion intake of residues (DynamicCROP, 2021). As to toxicity impacts, it has to be noted that in USEtox databases there is a lack of information about the impact of some inorganic and organic compounds used frequently in agriculture, hence more research is needed in this aspect.

As Notarnicola et al. (2022) claim, available datasets for agri-food processes may not be fully representative of the site-specificity of the food product under examination. There is thus a compelling need to develop country-specific datasets for fertilisers, pesticides and growth regulators. Those authors found that, specifically for citrus, no datasets concerning the production of Italian agricultural inputs are included within inventory databases. The lack of inventory datasets for the production of plant protection products, biological control agents, and organic fertilisers, key for organic agriculture, is highlighted in Montemayor et al. (2022), who make recommendations and give examples on how to create LCIs for these products. As to manure allocation, these authors recommend following the guidelines from the Livestock Environmental Assessment and Performance (LEAP) Partnership by the Food and Agriculture Organization of the United Nations (FAO, 2016a, 2018).

As regards water inventory, the optimum is to gather primary data of the water used for irrigation (e.g. water bills, water flow meters) although water balances, accounting for the soil and climate specificities, are also recommended when primary data is not available. In addition, irrigation infrastructure and the origin of the water used (surface, underground) should be detailed in LCAs. Payen et al. (2018) make a thorough review of models for field water flows for agricultural LCAs and recommend the CropWat model (Smith, 1992) as it allows actual crop irrigation requirements to be accounted for. The AquaCrop model (FAO, 2016b), the updated version of CropWat, is also highlighted by Payen et al. (2018) as an interesting alternative since it accounts for possible water, salinity, and nutrient stresses, which can affect crop growth and, thus, water flows. Its main weakness as regards the purpose of this research is that it still does not incorporate the modelling of perennials.

It must be borne in mind that two types of water can be defined, water extracted, that is the total amount of water withdrawn from water bodies (e.g. irrigation water, or water used for cleaning), and water consumed, which is the amount of water that the watershed of origin loses (Huijbregts et al., 2016). For the farming stage, water consumed by the crop corresponds to the evapotranspirated water without taking into account the rain water or the soil water capacity. Evapotranspiration depends on parameters such as climatic conditions, type of crop and its phenological stage. Water consumption is crucial data in the inventory and can be obtained by performing measurements on the field, using national databases or applying the method proposed by Allen et al. (1998).

4.3. Impact assessment

Most of the literature reviewed corresponds to multi-indicator studies. In fact, these are preferable as a means of reproducing the effect on the environment in a more representative way and identifying and preventing the burden-shifting between life cycle stages and between environmental indicators (Espadas-Aldana et al., 2019). This review shows that impact categories, such as climate change, eutrophication, acidification, or those that are toxicity related, are relevant in the LCAs of agri-food products. Notwithstanding, to regionalise the impact calculation, the recommendations are to apply global spatialized models that are regionally applicable and aligned with the geographic scope of the study and to apply continent- and country-level aggregated characterisation factors whenever possible (Patouillard et al., 2018). IMPACTWorld+ (Bulle et al., 2019) and LC-IMPACT (Verones et al., 2020) are recently developed regionalised methods. IMPACT World+ (Bulle et al., 2019) is a midpoint-damage framework that allows calculating midpoint indicators and endpoint indicators based on damage to three areas of protection (human health, ecosystem quality, and natural resources), together with two areas of concern related to water and carbon. In addition, the LC-IMPACT method (Verones et al., 2020) provides characterisation factors at the damage level for 11 impact categories related to the three areas of protection.

In addition, there are two impact categories that are relevant in agricultural LCAs and are barely assessed, namely water scarcity and biodiversity loss. Water scarcity, due to the intensive use of irrigation in citrus fruit crops and the fact that they are often located in contexts of water stress, should be systematically assessed. Recently, the working group of the UNEP-SETAC Life Cycle Initiative proposed the AWARE -Available WAter Remaining- method (Boulay et al., 2018), based on the quantification of the relative available water remaining per area once the demand of humans and aquatic ecosystems has been met. This method considers different spatial and temporal resolutions, proposing monthly characterisation factors, with a spatial specificity that reaches the country and even the basin level. In addition, different uses of water are considered as characterisation factors for both agricultural and non-agricultural activities are presented.

Although the impact on biodiversity is currently a relevant issue (e.g. the biodiversity strategy of the European Union for 2030) historically it was not considered, since agricultural lands were managed as industrial production sites (Notarnicola et al., 2017). In this regard, UNEP's consensus method uses the characterisation factors developed by Chaudhary and Brooks (2018), which enables to discern the damage caused by three levels of intensity within a particular broad land-use type, including three management regimes, as well as to contemplate taxon affinity to different types of land-use intensity. With regard to the toxicity of pesticides, the chosen characterisation method is important, as toxicity-related impacts vary significantly among them (Parajuli et al., 2019). The USEtox 2.0 model (Fantke, 2017) is the scientific consensus model endorsed by the UNEP/SETAC Life Cycle Initiative and the most up-to-date. Nevertheless, it must be pointed out that the terrestrial and marine ecotoxicity characterisation factors are still missing, and food ingestion is not accounted for in the human toxicity model.

4.4. Results interpretation and policy implications

LCAs have been used to support evidence-based policymaking, for instance, to inform consumers by comparing the environmental impacts of food supply, to communicate about the impact of mitigating interventions, as scientific basis for policies on products design, and to monitor sectoral progress towards sustainable development goals (Gava et al., 2020; Sala et al., 2021). In addition, Gava et al. (2020) point out how LCA studies could be beneficial for policymaking related to agricultural sustainability and food security. According to those authors, LCA can act as information providers, that is, by bringing new knowledge about the impacts concerning existing or novel products, they can highlight specific parameters in which policy measures (e.g. taxes or subsidies) or environmental performance standards could be sustained, or they can act as passive regulators, that is, helping to decide among various mitigating alternatives.

In this context, several barriers are still hindering the use of LCA in policy-making. Among them, Sala et al. (2021) highlight the lack of widespread technical knowledge on LCA, the lack of trust in the LCA process and results, and the need for verification of LCA results by surveillance authorities. Especially, weaknesses in the interpretation phase may contribute to reducing the trust of policymakers in LCA (Agostini et al., 2020). All this reinforces the need to develop harmonised methodologies, as well as an appropriate quantification of the uncertainty of the results to get grounded and representative results.

The goal of most of the reviewed articles is to provide information (see Tables 1 and 2). In addition, many of the articles reviewed propose measures to mitigate the identified hotspots, from more basic approaches, i.e., listing them, establishing theoretical scenarios that contemplate various alternatives or even making practical approximations. As well, comparisons between production systems are carried out, mainly between organic and conventional production. Along these lines, when performing a comparative analysis, it is crucial to assess whether the differences identified are significant or not, as noted in section 3.2.4.1. This can provide a consistent base to support decisions at the governmental level on which productive system to support. Hence, harmonisation is also required in this regard, by boosting the application of statistical inferential analysis (Grant et al., 2016; Sinisterra-Solís et al., 2020).

The Ecolabel Regulation (EC, 1992, 2010) constitutes a relevant application of LCA in policies establishing a voluntary eco-label award scheme to promote products with reduced life cycle environmental impacts (Sala et al., 2021). Through the use of these environmental certifications and labelling schemes, a reduction in the asymmetry of information from business to consumer can be achieved. In fact, governments and non-governmental organizations are nowadays fostering the use of ecolabels. This can be seen in the emergence of methods to measure the environmental performance of products and organizations, e.g. Environmental Product Declarations (EPD, 2022), and Product Environmental Footprint (EC, 2021), among others.

To give a holistic vision of the impacts of agricultural products, integrating the three dimensions of sustainability (environmental, economic, social) in a single index is recommended, as well as combining LCAs with other indicators, such as thermodynamic-based measures like exergy (Aghbashlo et al., 2021). This can be achieved by adopting a system thinking approach when performing interventions, helping to address multiple dimensions of sustainability at the same time. In this aspect, governments are key players in the adoption of macro-scale interventions and supporting research activities concerning progress monitoring.

5. Conclusions and future directions

Based on an extensive descriptive and critical literature review, this study analyses the state of the art of LCAs in citrus fruits. It identifies and discusses trends among the key methodological choices and makes recommendations for the development of a framework for a harmonised application of LCA on citrus fruits, which will help LCA practitioners to assess the impacts of fruit production in general. In addition, those system stages contributing the most to the environmental impact categories were identified, aiming to help farmers and managers of agri-food businesses to understand where the main changes should be monitored or implemented.

The production of agricultural inputs and their on-field emissions are the main critical points detected in the reviewed articles, hence, efforts should be directed towards the selection of more environmentally friendly products and to optimise their application rate. From this review, several recommendations and research gaps arise. Firstly, studies into the early stages of citrus fruit production (e.g. nursery stage) from primary data and with an exhaustive description of the inventory used are encouraged to determine whether they should be included or not. Farm representativeness, both temporal and spatial, stands out as a critical issue when assessing the regional production of citrus fruits. This implies improving farm sampling procedures or giving a rationale when only one farm is chosen instead and also increasing the number of years to be assessed to at least four. Regarding the life cycle inventories, the necessity to develop regionalized datasets of agricultural inputs is highlighted, also including specific inputs for organic production. As well, site-specific modelling tools for on-field emissions are crucial to obtain robust results that reflect local conditions; thus, the use and development of Tier 2 and Tier 3 methods should be fostered instead of Tier 1, although more research is needed to fit the models to local conditions. The need to improve water inventories, including not only the water withdrawal but also the water consumption, always accounting for regional representativeness, is highlighted. Farmers are encouraged to monitor the nutrient levels in their soils, which will allow for optimizing emission models while helping to adjust the dose of fertilisers applied.

As to the impact categories, the need to include water scarcity, and impacts on biodiversity in citrus fruits studies is highlighted together with the use of regionalised impact characterisation methods. The representation of uncertainty in the results of citrus fruits LCA studies is recommended, given the characteristics of agri-food processes, which are highly dependent on climate, soil type, farming practices, and other interrelated factors. As well, the use of statistical methods when comparing the impact results of different systems stands out as a key aspect to ensure the consistency of the obtained results.

This study contributes to the harmonisation of citrus fruits LCA studies as a first step that paves the way for the promotion of a more responsible and sustainable citrus fruit production. LCA studies can help product differentiation and its incorporation in new and more exigent international markets. Studies in other citrus-producing regions worldwide should be fostered to acquire a broader picture of the global environmental impacts and its site-specific dependency thus helping policymakers to develop evidence-based environmental policies.

Declaration of competing interest

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

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