Life cycle assessment of aquaculture bivalve shellfish production — a critical review of methodological trends

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Abstract

Purpose The increase of shellfish production has raised environmental concerns, i.e., enrichment and redistribution of nutrients and energy consumption. Efforts assessing the environmental burdens arising from the expansion of shellfish production have been made using the life cycle assessment (LCA) methodology. Although LCA has been extensively applied and reviewed in aquaculture systems, shellfish production remains scarcely studied. The objective of this review is to identify methodological trends, highlight gaps and limitations, and provide guidelines for future studies.

Methods A systematic literature review was applied to scientific studies published up to 2021. A total of 13 documents were shorted by abstract and full text-screening. Literature meeting the inclusion criteria were further analyzed in six different aspects of a LCA (functional unit, system boundaries, data and data quality, allocation, impact assessment methods, interpretation methods). Discussion and guidelines are provided for each reviewed aspect.

Results and discussions Shellfish LCAs differ considerably from other aquaculture studies mainly because shellfish avoids the allocation of impacts derived from the production of fishmeal. Co-products are present when the shellfish is processed, e.g., in canned products. Furthermore, shellfish studies do not take into account the positive credits from the removal of nutrients from the ecosystems and from the valorization of the shellfish waste (shell and organic remains). Limited information was found for countries outside Europe and species different from mussels. Despite the variability on goals and scopes of the studies, methodological trends were found. The local impacts of the shellfish with the farming area and the impacts on biodiversity have not been included into the studies.

Conclusions and recommendations Effort should be made in providing the data associated with the fore-background system within the studies in order to improve transparency and to allow the reproduction of the results. Information regarding the natural condition of the cultivation area should be provided as the shellfish production depends mainly on non-anthropogenic conditions. Application of biodiversity assessment methodologies should be encouraged, despite their limitations.

Keywords Aquaculture · Bivalve · LCA · Life cycle assessment · Nutrient removal · Review · Shellfish

1 Introduction

Seafood from aquaculture has increased its share of the total aquatic food consumed globally from 26% in 2000 to 46% in 2018 (FAO 2020b). Such increase may be attributed, among others, to the increased demand not only for protein for human consumption but also for derivate co-products used

in several industries, e.g., feed, fuels, cosmetics, nutraceuticals, and pharmaceuticals (Naylor et al. 2021), the human population growth, the development of more efficient farming techniques, and the fact that seafood from wild fisheries production is limited (FAO 2020b). Among the many species farmed commercially, bivalve shellfish, hereafter referred to as “shellfish,” have experienced an increase in global annual harvest of over 90% from 8.3 million tonnes to 15.9 million tonnes between 2000 and 2018 (FAO 2020b). Mussels in particular have seen the greatest increase in production compared to the other shellfish species. Within the last two decades, total mussel production has increased by 62%, followed by oysters (60%); scallops and pectens (46%); clams, cockles, and arkshells (42%); freshwater mollusks

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(12%); and finally pearls, mother of pearl, and shells (3%) (see Fig. 1). Shellfish farming is considered sustainable and environmentally-friendly compared with other aquaculture species (Brigolin et al. 2009). It provides multiple ecosystem services, e.g., eutrophication control (Lindahl et al. 2005), increase of water transparency, and the promotion of macrophyte growth (positive water quality indicators in the Water Framework Directive (WFD)) (Schröder et al. 2014), as well as promotion of habitat to different species, e.g., algae, worms, snails, crustaceans, and fish (Ysebaert et al. 2009).

Aquaculture is a core element of the EU Blue Growth strategy (European Commission 2012). Shellfish farming as a part of the mentioned strategy has been catching the attention of government agencies and entrepreneurs since it may remediate the eutrophication issues in degraded water bodies, such as the Baltic Sea (Lindahl et al. 2005; Lindahl and Kollberg 2009; Petersen et al. 2014). It is recognized as a prominent cost-effective mechanism to remove nutrients compared with other abatement measures (Gren et al. 2009). Therefore, a variety of projects such as the Baltic EcoMussel project (European Commission 2013), the Baltic Blue Growth project (BBG) (European Commission 2016), and the BONUS OPTIMUS project (European Commission 2017) have been launched to quantify the possible benefits of farming shellfish in the Baltic Sea.

However, shellfish may also have potential negative environmental effects, e.g., ecosystem degradation (Danovaro

et al. 2004), energy consumption, climate change, and other effects (Aubin et al. 2018; Iribarren et al. 2010a, b, c; Lourguoui et al. 2017; Tamburini et al. 2020) across the entire life cycle production (seeding, collection, growing, harvesting, processing, and waste management). A common tool to assess the environmental impacts also for aquaculture systems is the life cycle assessment (LCA) (Bohnes et al. 2019). The LCA-methodology has been extensively applied in several aquaculture systems (e.g., cages, ponds, recirculating aquaculture systems (RAS)) and species (e.g., salmon, tilapia, catfish, and shrimps) from the beginning of the twenty-first century (Aubin 2013; Bohnes et al. 2019; Bohnes and Laurent 2019; Cao et al. 2013; Henriksson et al. 2012; Pahri et al. 2015; Parker 2012). For example, Henriksson et al. (2012) reviewed a dozen LCAs from a methodological perspective (including three LCAs on shellfish). Similarly, Parker (2012) reviewed three shellfish studies from 62 reviewed documents (3/62), Aubin (2013) (3/50), Cao et al. (2013) (1/12), Bohnes and Laurent (2019) (7/65), and Bohnes et al. (2019) (7/65) to identify patterns in objectives, parameters, and methodologies in seafood. Finally, Pahri et al. (2015) analyzed the influence of natural and anthropogenic factors on aquaculture systems by reviewing 15 scientific papers (one shellfish).

Above-mentioned literature review articles represent a large sample of assessments covering several methodological aspects of a LCA such as functional unit, system

Fig. 1 Shellfish production by families during the period 2000–2018. Source: FAO (2020a)

boundary, data and data quality, allocation methods, environmental assessment method, and result interpretation. Detailed analysis for shellfish aquaculture, however, is still missing. Unlike other aquaculture species, shellfish are filter-feeders that feed on the phytoplankton and organic matter present in the water column (Brigolin et al. 2009). This fact plays a key role in the relatively low environmental impact of mussel farming compared with other animal production processes since feed production is responsible for about 56% of the total impacts associated with climate change (CC) in seafood production (Aubin et al. 2018; Bohnes et al. 2019). Additionally, they do not require the use of antibiotics to fend off diseases (Iribarren et al. 2010b). The use of antibiotics potentially creates antimicrobial resistance, an issue that has been acknowledged as one of the biggest threats to global health and food security by the World Health Organization (WHO 2020).

The goals of this study are to conduct a systematic and critical review of existing shellfish LCAs to (1) provide a mapping of studies published up to 2021, (2) identify most common modeling practices and environmental impacts studied, (3) identify hot spots of environmental impacts along the life cycle of the shellfish, and (4) deduce recommendations for future LCA developments in this field of research. Our results are relevant to several stakeholders. Policymakers and entrepreneurs will gain knowledge of the potential environmental impacts and their hot spots in shellfish aquaculture from a broader holistic view that includes the shellfish aquaculture supply chain. Moreover, practitioners will gain insight into the main methodological aspects as well as the role of the nature-related aspects of farming influencing the environmental footprint.

2 Material and methods

This work comprises a systematic literature review of empirical studies conducting an LCA on shellfish aquaculture. It is based on the process described by Khan et al. (2003) to conduct a systematic review, which includes defining review questions, identifying relevant work, selecting criteria, synthesizing the evidence, and interpreting the findings.

A literature search of international publications in English was conducted in June 2021 using Scopus. The search string was iteratively tested and refined so that it would be both synthetic and comprehensive. The final search string was as follows:

TITLE-ABS-KEY (("Mussel" OR "bivalvia" OR "oyster" OR "ostrea" OR "Clams" OR "scallop" OR "shellfish") AND ("life cycle assessment" OR "life cycle analysis" OR "LCA" OR "life cycle inventory" OR "LCI" OR "material flow analysis" OR "sustainability assessment")).

This resulted in 83 documents, which were further screened based on the document's title, authors, and abstract. To be suitable for inclusion in the review, the document had to be (1) a scientific article, book or book chapter, or report and (2) a study conducting a systematic LCA of shellfish. During this process, documents were rejected because they did not comply with the inclusion criteria. The final sample of articles included 13 documents (10 documents from the initial search in Scopus and three documents added after cross-citation analysis of the documents).

3 Results

This section presents the statistical analysis and the main findings associated with different methodological aspects. As done by Henriksson et al. (2012), results are divided into the functional unit, system boundary, data and data quality, allocation methods, environmental impact assessment method, and interpretation method.

3.1 Statistical analysis

A total of 13 documents were collected from which 10 were peer-reviewed articles, one book chapter, and one reports. Geographical distribution of the studies reveals important discrepancies (see Table 1). The majority of the studies (85%) are located in Europe (four in Spain, three in Italy, one in France, and one in Sweden, Norway, and Scotland, respectively); only one study is located in Africa (Algeria) and South America (Brazil), respectively. About the year of publication, 30% of the sample (four studies) were published in 2010; in addition, 15% (two documents) were published in 2020. The rest of the sample were published between 2012 and 2019.

Regarding the type of species, mussels are the most studied (71% of the total sample), from which 60% studied the Mediterranean mussel (*Mytilus galloprovincialis*) and 40% the Blue mussel (*Mytilus edulis*). Three documents studied oysters (*Crassostrea gigas*) and one clams (*Ruditapes philippinarum*) (see Table 1).

The most used technique for the shellfish culture is rafts (28% of the sample four studies); furthermore, longlines represent 21% of the sample (three studies). Beds are used in one study, while nets and bouchots (wooden pilings) are each used once. It is worth noting that the studies associated with the rafts provided by Iribarren et al. (2010a, b, c, d, e) were developed almost with the same data set and similar methodological approach. Moreover, three studies do not provide information regarding the technique used (Alvarenga et al. 2012; Fry 2012; Ziegler et al. 2013).

The most used impact assessment method is the CML method (53% of the sample or seven documents) of which

Table 1 Reviewed documents

Study	Location	Shellfish	Species	Technology	FU	Methodology	Impact category	Software	Type document
Iribarren et al. (2010c)	Spain, Galicia	Mussel	<i>Mytilus galloprovincialis</i>	Raft	89.74 tonnes fresh mussel	CML 2000	CC, E, A, POF, OD, MET, HT, AD, FET, TET	SimaPro 7.0	Book chapter
Iribarren et al. (2010b)	Spain, Galicia	Mussel	<i>Mytilus galloprovincialis</i>	Raft	1 kg fresh mussels, 1 kg canned mussel	CML 2001	CC, E, A, POF, OD, MET, HT, AD, FET, TET	SimaPro 7.0	Article
Iribarren et al. (2010a)	Spain, Galicia	Mussel	<i>Mytilus galloprovincialis</i>	Raft	100 tonnes mussel shell, 100 tonnes mussel meat	CML 2001	CC, E, A, POF, OD, MET, HT, AD, FET, TET	SimaPro 7.0	Article
Iribarren et al. (2010d)	Spain, Galicia	Mussel	<i>Mytilus galloprovincialis</i>	Raft	100 kg cultivated mussels	CML 2001	CC, E, A, POF, OD, MET, HT, AD, FET, TET	SimaPro 7.0	Article
Alvarenga et al. (2012)	Southern Brazil, Florianopolis	Oyster	<i>Crassostrea gigas</i>	ND	1 kg of oyster	Ecoindicator 99-H	HH, EQ, RD	SimaPro 7.3.0	Article
Fry (2012)	Scotland	Oyster	<i>Crassostrea gigas</i>	ND	1 kg of oyster	PAS 2050	CC	ND	Report
Fry (2012)	Scotland	Mussel	<i>Mytilus edulis</i>	ND	1 ton fresh mussel	PAS 2050	CC	ND	Report
Ziegler et al. (2013)	Norway	Mussel	<i>Mytilus edulis</i>	ND	1 kg of fresh mussel	IPCC 2007	CC	SimaPro 7.3.2	Article
Spångberg et al. (2013)	Sweden, Kalmar County	Mussel	<i>Mytilus edulis</i>	Nets	1 kg nitrogen, 0.88 kg phosphorus, 225 kg CaO	CML 2001	CC, E, A, EU	SimaPro 7.3.0	Article
Lourguoui et al. (2017)	Algeria, Ain Chrob	Mussel	<i>Mytilus galloprovincialis</i>	Long-line	1 ton fresh mussel	CML 2000; CED V 1.08	CC, E, A, EU	SimaPro 8.0.3.14	Article
Joël Aubin et al. (2018)	France, Mt.-St. Michel bay	Mussel	<i>Mytilus edulis</i>	Bouchot	1 ton fresh mussel	CML 2000; CED V 1.03; Aubin (2009)	CC, A, E, EU, WD	SimaPro 7.3.0	Article
Tamburini et al. (2019)	Northern Italy, Po delta	Oyster	<i>Crassostrea gigas</i>	Long-line	1 kg fresh oyster	Ecoindicator 99-H; ReCiPe midpoint (H) v.1.12	HH, EQ, RD	OpenLCA 1.8.0	Article
Turolla et al. (2020)	Northern Italy, Po delta	Clam	<i>Ruditapes philippinarum</i>	Bed	1 kg fresh clam	ReCiPe midpoint (H) v.1.12	CC, E, A, POF, OD, MET, HT, FD, WDP	OpenLCA 1.8.0	Article
Tamburini et al. (2020)	Northern Italy, Po delta	Mussel	<i>Mytilus galloprovincialis</i>	Long-line	1 kg fresh mussel	CML 2000	CC, E, A, HT, AD, OD, POF, MET	OpenLCA 1.8.0	Article

Climate change (CC), eutrophication (E), acidification potential (A), water dependency (WD), energy use (EU), human health (HH), ecosystem quality (EQ), resource depletion (RD), photochemical oxidant formation (POF), ozone layer depletion (OD), marine aquatic ecotoxicity (MET), human toxicity potential (HT), fossil depletion (FD), water depletion (WDP), abiotic depletion (AD), fresh water aquatic ecotoxicity (FET), terrestrial ecotoxicity (TET), freshwater eutrophication (FEU), marine eutrophication (MEU), cumulative energy demand (CED)

half used CML 2000 and the other CML 2001 (see Table 1). ReCiPe midpoints and ecoindicator 99 endpoints are used twice each. Finally, the most widely used LCA software is SimaPro (69% of the samples).

3.2 Functional unit (FU)

The most common FU used by the reviewed studies (see Table 1) is 1 weight unit (ton or kg) of packaged shellfish. Aubin et al. (2018) and Tamburini et al. (2020) defined the FU as 1 ton of packaged mussels. Alvarenga et al. (2012) and Tamburini et al. (2019) defined the FU as the provision of a single meal (1 kg of packaged oysters), and Turolla et al. (2020) used 1 ton of packaged clams ready to cook as FU. Lourguioui et al. (2017), Fry (2012), and Ziegler et al. (2013) studied the environmental impacts of 1 ton or kg of fresh mussel without any packaging activities associated.

In contrast to the former Iribarren et al. (2010c) defined the FU as the annual mussel production of a conventional raft (89.74 tonnes). Iribarren et al. (2010b) labeled the FU as the industrial processing and consumption of 1 kg of fresh mussel and 1 kg of canned mussel flesh. Iribarren et al. (2010d) defined the FU as 1 kg of edible mussel flesh to compare two industrial processing paths for fresh mussel (freezing of boiled mussels and canning of boiled mussels). Iribarren et al. (2010a) defined the FU as the production of 65 tonnes of calcium carbonate (CaCO_3) and 278 tonnes of mussel pâté for the mussel shell and the mussel organic remains, respectively.

Differently, Spångberg et al. (2013) studied the potential of mussels to replace fertilizer assuming that the mussel fulfill multiple functions: (1) fertilizer value, (2) liming effect, and (3) eutrophication reduction, thus defining the functional unit as the production and application of 1 kg nitrogen, 0.88 kg of phosphorus, and 225 kg CaO on arable land simultaneously.

3.3 System boundary

A significant sample (77% of the reviewed studies) applied a cradle to gate boundary with the inclusion of the cleaning and packaging activities within or after the farming stage according to the particular conditions of the market assumed, mainly marketed as fresh meat. Turolla et al. (2020), Tamburini et al. (2020), Tamburini et al. (2019), Ziegler et al. (2013), and Fry (2012) included purification (i.e., depuration) of the shellfish (seawater with/without chemicals) and packaging activities after harvesting. Additionally, Iribarren et al. (2010c) assume for the mussels to be washed with sea water and packed in sacks prior to industrial processing. Lourguioui et al. (2017) in contrast do not include the packaging stage.

On the other hand, Aubin et al. (2018) included the activities of purification (seawater and without chemicals), separation from ropes and nets, cleaning and brushing, and packaging prior to being sold to the consumers and include the end of life (EoL) of the mussels after the consumption stage.

Iribarren et al. (2010b) initially focus on the industrial processing of two products (fresh mussel and canned mussel) with the inclusion of the consumption stage and EoL. Purification (water with chlorine gas) is performed in this case only for the alternative of fresh mussel. Moreover, the farming stage is aggregated with information obtained from Iribarren et al. (2010c). Similarly, Iribarren et al. (2010a) estimated the environmental burdens associated with the use of mussel shell and the organic remains to produce CaCO_3 and mussel pâté, respectively.

From the cradle to grave perspective (23% of the reviewed studies), Iribarren et al. (2010d) combined the results obtained in Iribarren et al. (2010a, b, c) to study the potential impacts arising from the mussel sector in Spain. Spångberg et al. (2013) studied the potential of mussel to replace conventional fertilizer and to uptake nutrients from the sea. For doing so, the authors included within the system boundary the farming stage, the transport to the farm, the fertilizer production (composting or crossing), the spreading of the fertilizer, and the EoL of the mussel farming materials, the main materials in fishing vessels, machinery, and the rubber (Spångberg et al. 2013). Similarly, Alvarenga et al. (2012) assessed the environmental impacts of oyster farming in Brazil including the farming, packaging, consumption, EoL of the shell, and the transport between stages. The study aimed to particularly assess the potential benefit of shell waste to be used as an alternative to the conventional mining extraction of CaCO_3 .

Capital goods are included into the studies with different levels of coverage mainly due to the lack of reliable data. For example, Alvarenga et al. (2012) included only capital goods associated with the farming stages of the oysters; likewise Turolla et al. (2020), Tamburini et al. (2020), and Tamburini et al. (2019) applied a similar approach and additionally included the technical clothing and gear required for the operation of the farm, though with the exclusion of transport and the EoL stage. Furthermore, Lourguioui et al. (2017) only included the capital goods associated with the construction of the farm (e.g., ropes, floats, anchors), conversely the boat, and machinery involved in other stages were excluded. Similarly, Iribarren et al. (2010c) included the capital goods associated with the farming (vessels included) and the EoL of some components, e.g., cotton, nylon, polyethylene, and polypropylene. Moreover, Aubin et al. (2018) included the production of equipment, e.g., boat and machinery, while EoL is not mentioned. Finally, Spångberg et al. (2013) included the production and maintenance of capital goods associated with

mussel farming and all agricultural activities involved in the production of the fertilizers.

Few studies (38%) account for the uptake of nutrients, e.g., carbon (C), nitrogen (N), and phosphorus (P). Lourguioui et al. (2017) and Aubin et al. (2018) used a model developed by Brigolin et al. (2009) to estimate the nutrient fluxes and take into account the positive environmental impacts on eutrophication associated with the uptake of nutrients during the entire growing cycle. Similarly, Aubin et al. (2018) estimated the C fixed in the shell and the C content on the wooden stakes to account for benefits of sequestrating of C. Furthermore, Turolla et al. (2020) estimated the positive impact associated with the C, N, and P fixed in the clam shells using data from Zan et al. (2014) and the formula proposed by Ray et al. (2018) to estimate the carbon dioxide (CO₂) release during the formation of the shell. Finally, Spångberg et al. (2013) and Fry (2012) included positive impacts associated with the sequestration of N, and P. Fry (2012) included positive credits for the C, although both authors do not specify the method used for the calculation.

3.4 Data and data quality

The foreground system is mainly modeled with primary data, e.g., interviews and questionnaires. Moreover, data representativeness, a criterion associated with the quality of the data, is documented to different extent. While in Lourguioui et al. (2017) data gathered represent around 30% of the national Algeria's mussel production in 2013 (150 tonnes); for the other studies, quantities are mentioned but the representativeness is not explicitly declared. For example, Turolla et al. (2020) interviewed 119 clam farmers representing 988 tons of the harvest of the region. Aubin et al. (2018) collected data from producers with a total yield between 300 and 450 tonnes per year in 2012 and data for one producer of seeds, since this activity was initially not included in the study. Additionally, some authors do not declare the type and quality of data provided; e.g., Tamburini et al. (2020) gathered data from different sources, interviews, and questionnaires for 30 mussel farmers representing about 75% of the overall local production, and data based on professional experience of the authors. Tamburini et al. (2019) used information based on personal experiences and complemented it with interviews of local farmers accounting for a yield of 8 tonnes per year.

Data used in Iribaren et al. (2010a, b, c, d) to model the supply chain of the mussel industry in Spain was obtained from the skippers of 22 auxiliary vessels in charge of 80 rafts (7,180 tonnes of total production), and different companies, such as a dispatch center, a canning factory, a factory that processes the mussel shell, and a pâté factory. While presumably the data provided comes from the most important companies in the supply chain, representativeness of the data is not declared. Moreover, complementary

data were collected from the suppliers of additional materials, e.g., engines, equipment, and hydraulic cranes. Finally, statistical data and primary data were collected to model the consumption stages.

Alvarenga et al. (2012) collected primary data for the farming stage based on the production of oysters in 2006 with quantity and representativeness left undeclared. Additionally, secondary data was used to model the shell processing. Finally, Spångberg et al. (2013) do not specify the type of data collected. The authors refer to Kalmar county (Sweden) as location and only mentioned a production of 150 tonnes per year. Moreover, the authors state that data used for the farming stage was based on personal communications with retailers of cultivation materials.

The background system in all the reviewed studies was modeled with ecoinvent database in their different versions: Turolla et al. (2020), Tamburini et al. (2020), and Tamburini et al. (2019) version 3.6; Aubin et al. (2018) version 3.1; Lourguioui et al. (2017) version 3; Alvarenga et al. (2012) version 2.2; Spångberg et al. (2013) version 2.1; and Iribaren et al. (2010a, b, c, d) and Ziegler et al. (2013) version 2.0. Fry (2012) does not specify the version used.

3.5 Allocation methods

Most of the studies reviewed (78%) do not apply any allocation method, mainly because the functional unit defined by the different authors is the production of 1 unit weight (shell included) of fresh shellfish (Alvarenga et al. 2012; Aubin et al. 2018; Iribarren et al. 2010c; Lourguioui et al. 2017; Tamburini et al. 2019, 2020; Turolla et al. 2020). It is worth noting that Aubin et al. (2018) in a second step re-defined the functional unit as 1 ton of edible protein to compare the results with other animal products. Therefore, the authors assume that 27% of the wet weight (shell included) is flesh of which 11.9% is protein (Aubin et al. 2018).

On the other hand, Iribarren et al. (2010b) performed mass allocation for the industrial processing of the mussel in the cleaning and purification stage given the fact that the dispatch center not only processes mussels. For the remaining industrial activities allocation was not needed because the canned mussel production has its own production line and, therefore, processes involved refer only to the mussels.

The substitution approach (also known as avoided burden approach) was applied in Iribarren et al. (2010a). On the basis that the CaCO₃ and mussel pâté produced from the shell and the organic remains avoid additional production of CaCO₃ and mussel pâté, respectively. Similarly, Spångberg et al. (2013) applied the substitution approach for the production of fertilizer from mussel biomass (flesh and shell).

3.6 Environmental impact assessment method

Regarding the environmental impact categories included, all the studies use climate change (CC), eutrophication (E), and acidification (A). Turolla et al. (2020), Tamburini et al. (2020), and Iribarren et al. (2010a, b, c, d, e) included the photochemical oxidant formation (POF), ozone layer depletion (OD), marine aquatic ecotoxicity (MET), and human toxicity (HT). While Turolla et al. (2020) included fossil depletion (FD) and water depletion (WDP) Iribarren et al. (2010a, b, c, d, e) included abiotic depletion (AD), fresh water aquatic ecotoxicity (FET), and terrestrial ecotoxicity (TET). Alvarenga et al. (2012) and Tamburini et al. (2019) used eco-indicator 99-H (hierarchical version) to account for the impact in human health (HH), ecosystem quality (EQ), and resource depletion (RD). In a second step Tamburini et al. (2019) use the ReCiPe midpoint (H) to perform comparisons with other aquaculture products.

Water dependency (WD) and energy use (EU) were used in some studies (35%). Lourguioui et al. (2017) and Aubin et al. (2018) used the Cumulative Energy Demand (CED) method (Frischknecht et al. 2007) to account for the energy consumption. Moreover, Aubin et al. (2018) used the method developed by Aubin et al. (2009) to assess WDP impact. Additionally, Spångberg et al. (2013) calculated the EU but no clear reference of the method used is provided. Turolla et al. (2020) used the WD indicator provided by ReCiPe.

3.7 Result interpretation

A few studies (23% of the total) only applied a contribution analysis to identify the most significant stages or materials influencing the environmental impacts of the system under study (Fry 2012; Iribarren et al. 2010b, d).

The environmental impacts associated with the farming stage are mainly related to energy consumption. Lourguioui et al. (2017) suggest that around 57.6% (CC) and 59.7% (A) of the total impacts are linked to the energy consumption (fuel and electricity), from which the fuel consumption of the vessels represents 19.5% (CC) and 27.9% (A) of the total impacts; conversely, the electricity consumed during the farming, washing, cleaning, and grading represents 38.1% (CC) and 31.8% (A) of the total impacts. Moreover, the longlines and the buildings required for the processing of the mussels each produce around 20% of the burdens in the impact categories studied. Similar ranges are provided by Iribarren et al. (2010c) where the electricity consumption accounts for between 15% (ODP) and 80% (CC) of the impacts and the fuel consumption produces around 80% (OD) and 5% (POF) of the total impacts across the different impact categories analyzed. Furthermore, Aubin et al. (2018) suggest that the fuel consumption represents between 38% (CED) and 54% (A) of the burdens across the impact

categories studied. While in Turolla et al. (2020) the values range between 30% (CC) and 80% (FD), and in Tamburini et al. (2020), the values range between 50% (CC) and 88% (OD) of the total impacts across the different categories.

Similar results can be found for the industrial processing stage. Iribarren et al. (2010b) suggest that the electricity represents between 5% (EP) and 95% (POF) of the total impacts across the different environmental impact categories, whereas the chloride used during the purification stage accounts for around 5% (A) and 35% (ODP) of the total impacts. Similar results can be found for the valorization of the shell and the organic remains. Iribarren et al. (2010a) refer to an impact contribution 30% (AD) and 50% (A) for the energy consumption.

To assess the influence of data quality and the assumptions made on the results, 33% of the total studies conducted an uncertainty and a sensitivity analysis. Lourguioui et al. (2017) used a scenario analysis to study the impact of the mussel production under two scenarios (competitive and cooperative) for the post-harvesting stages. Similarly, Aubin et al. (2018) assessed the influence of different assumptions related with the credits granted for the uptake of C. Tamburini et al. (2020) evaluated three different assumptions: (1) substitution of an electric engine on the boat for a diesel engine, (2) replacing of 50% of the high-density polyethylene (HDPE) used with hemp textile yarn as the material for the socks, and (3) replacing 100% of the HDPE used with hemp textile yarn for the socks. Finally, Tamburini et al. (2019) studied the potential environmental benefits of producing the seeds in situ instead of importing them in France. All above mentioned studies conducted an uncertainty analysis for the foreground data.

Finally, Turolla et al. (2020) conducted only an uncertainty analysis, while Iribarren et al. (2010a, c), Spångberg et al. (2013), and Alvarenga et al. (2012) performed only a sensitivity analysis. Iribarren et al. (2010c) estimated the effects of a reduction of the energy demand for capital goods (reduction of 10%, 20%, and 50%) on the environmental impacts of the mussel farm. Iribarren et al. (2010a) studied different EoL paths for the mussel shell, e.g., incineration, landfill, and the production of CaCO_3 , and the mussel organic remains, e.g., production of fish meal and mussel pâté. Spångberg et al. (2013) assessed the influence of changes in materials and diesel consumption of the vessels. Moreover, the mussel fertilizer was compared with chemical fertilizer and meat meal. Finally, Alvarenga et al. (2012) evaluated the potential environmental impacts of different EoL paths for the oyster shell, e.g., landfill and production of CaCO_3 .

It is worth mentioning that regarding the EoL of the shell and organic remains, Iribarren et al. (2010a) suggest that the shell sent to a landfill disposal shows a better environmental profile compared with the incineration alternative. However,

landfill disposal does not offer marketable products. In contrast the valorization of the organic remains by producing mussel pâté is economically and socially favored and has a better environmental profile than the fish meal alternative (Iribarren et al. 2010a).

4 Discussion

This section discusses the main findings associated with above summarized methodological aspects of the LCA.

4.1 Functional unit

The functional unit follows the goal and scope of the study (ISO-14044 2006) and it can be understood as the bridge between the life cycle inventory (LCI) of the product system and the environmental impacts. Moreover, the functional unit allows for comparisons to be made between different product systems. Thus, different goals may require different functional units and consequently different system boundaries may be needed; e.g., if canned mussels ready to cook are chosen as a functional unit as done in Iribarren et al. (2010b), the boundaries under study should include the industrial processing of the harvested mussel, whereas the transport, EoL of the shell, and organic remains could remain as optional in accordance with the goal of the study. Similarly, if the pursued goal is to understand the environmental impacts associated with the farming stages, the activities associated with the post-harvesting stage, e.g., industrial processing, distribution, and consumption, may remain outside of the system boundaries and the functional unit should be in accordance, e.g., 1 kg harvested shellfish as done in Iribarren et al. (2010c).

A proper definition of the functional unit becomes critical when comparisons of farmed species or culture techniques are intended. For example, when shellfish is compared with other animal products, e.g., fish, the functional unit of 1 kg fresh shellfish, commonly used in the reviewed studies (see Table 1), does not provide a fair comparison between systems. This is mainly because shellfish is composed of two elements (meat and shell), from which, according with Aubin et al. (2018) for the case of the blue mussel (*Mytilus edulis*), the meat represents only 27% of the total weight. Therefore, when comparing the two different systems the result on average may be underestimated in favor of shellfish. In this case, a most suitable function should be 1 kg of edible meat. Furthermore, McAuliffe et al. (2020) argues that food-based LCAs should be assessed and compared on the basis of the true function of the food (provided nutrition). In this case, comparisons should be made in function of energy-based (kcal) or nutrition-based (protein) as done in Aubin et al. (2018). The authors compared the mussel and

scallop production, respectively, against other types of animal protein production (cow milk, chicken, pork, egg, beef, and fish from aquaculture) on the basis of their nutritional content. For doing so, e.g., Aubin et al. (2018) assumed that the mussel has 27% of flesh with a protein content of 11.9%. It is worth mentioning that compared with agricultural animal products (above mentioned) per ton of edible protein, mussel production is more energy demanding (between 225.27 and 335.36 MJ/Ton protein) than agricultural animal products (between 81.39 and 158.88 MJ/Ton protein) as mussels have a smaller yield in edible protein (Aubin et al. 2018). However, when climate change impact is compared, the result for the shellfish is much lower (296 kg CO₂ eq/ton protein) than for the other mentioned animal protein (between 19.53 and 83.15 Ton CO₂ eq/ton protein) studied by Aubin et al. (2018).

Comparisons between different systems become complex when the system under study is deemed to provide multiple functions rather than one. Spångberg et al. (2013) suggest that the mussel base fertilizer may supply different functions: fertilizer value, liming effect, and eutrophication reduction. In order to conduct comparison with conventional fertilizer the authors include a wastewater treatment plant to the conventional fertilizer system to equalize the nutrient removal function of the mussel. Although the study highlighted the importance of considering the multiple functions provided by mussels applied to agricultural land, the system expansions should be executed and treated with care and documented transparently.

It is recommended to use sensitivity analysis to properly assess the influence of the functional unit on the results as done in Iribarren et al. (2010d). Besides, in agreement with Henriksson et al. (2012) the functional unit should be carefully defined when comparisons between cultures are intended, as the definition of edible goods depends on cultural consumer preferences.

4.2 System boundary

To provide a comprehensive analysis of the environmental impacts associated with shellfish a full life cycle perspective is needed. Stages such as the industrial processing, transport, and consumption may increase the environmental burdens, while others like EoL may bring environmental benefits (Bohnes et al. 2019). This is particularly important for processed products with added value (Lourguioui et al. 2017). In such regards, few studies included the impacts associated with the supply chain beyond the cultivation stage. Similar results can be found for other types of aquaculture systems, e.g., fish and shrimp (Bohnes et al. 2019; Henriksson et al. 2012).

Industrial processing may add significant environmental impacts to the system depending on the type of product

produced, e.g., fresh mussel or canned mussel. Iribarren et al. (2010b, d) found that across impact categories the processing of fresh mussels is responsible for between 60% (FEP) and 95% (E) of the environmental impacts; in contrast, the canned mussel alternative is responsible for about 5% (MET) and 60% (E) of the impacts. In both cases farming is the second most significant stage and the consumption stage is not significant (Iribarren et al. 2010b). Differences between the two types of industrial processing are mainly related to the purification stages, while for the fresh mussel the purification is achieved with chlorine. In facilities with high energy consumption, canned processing is conducted in optimized facilities and does not require a purification stage (Iribarren et al. 2010b, d). It is worth mentioning that Iribarren et al. (2010b) provide the only study in which chlorine is used in the purification of the mussel. Similarly, Spångberg et al. (2013) suggest that the processing of the mussel into fertilizer represents more than 60% of the environmental impacts associated with CC; additionally, around 35% of the CC impacts are associated with the farming stage.

The EoL stage may have significant burdens across the different impact categories studied depending on the fate of the shells. Alvarenga et al. (2012) suggest that EoL may represent between 1 and 41% of the total impacts when the shell is disposed of in a landfill or processed into CaCO_3 , respectively. Similarly, Iribarren et al. (2010a) state that, across impact categories, the processing of shell into CaCO_3 may represent between 10 and 30% of the total burdens. Conversely, the mussel pâté may account for around 5% and 16% of the total impacts.

Capital goods represent a non-negligible impact across the different impact categories and stages. Iribarren et al. (2010c) demonstrate that capital goods associated with the mussel farming may represent between 21 and 103% of total impacts across the different environmental categories, including positive benefits. In contrast the capital goods associated with the industrial processing of the mussel may represent between 18 and 30% of the environmental impacts associated with the A, E, and CC environmental impact categories (Lourguioui et al. 2017). Results provided by Iribarren et al. (2010c) and Lourguioui et al. (2017) suggest that the environmental burdens arising from the capital goods used in the industrial processing stage are significantly lower but non-negligible to those associated with the farming stage.

Contrary to the farming stage in which the capital goods are to some extent included, the capital goods in the industrial processing stage are commonly excluded mainly due to the lack of reliable data and the widely spread assumption that in a long life span, commonly 20–25 years for industrial facilities and 30 years for vessels, the environmental impacts associated to capital goods become negligible (Iribarren et al. 2010a, b, c, d, e).

Effort should be made in expanding the system boundary beyond the farming stages and in gathering information about the capital goods. One alternative to solve the issues associated with the exclusion of processes or stages via cut-off is the environmentally extended input–output analysis (EEIOA) (Suh et al. 2004). The EEIOA is a top-down technique that uses monetary units to estimate environmental burdens across industry transactions (Suh et al. 2004) and is particularly useful to account for the impacts of flows for which no detailed data is available (Henriksson et al. 2012). In EEIOA the capital goods can be taken into account once the monetary expenditures associated with it are available. In any case, the system boundary as well as the main inclusions and exclusions should be in consistency with the goal of the study and clearly declared in order to add transparency and facilitate interpretations and comparisons between studies (ISO-14044 2006).

In contrast to other aquaculture species, shellfish are a buffer against eutrophication (Nizzoli et al. 2005). They are filter-feeders and fix nutrients, e.g., N and P in the flesh and shell during their outgrowing. While the uptake of nutrients from shellfish has been well documented (Lindahl et al. 2005; Lindahl and Kollberg 2009; Petersen et al. 2014), the accounting for this ecosystem service has not been fully explored in LCA studies as 23% of the sample include positive credits for the uptake of nutrients. See Aubin et al. (2018) (*Mytilus edulis*), Lourguioui et al. (2017), (*Mytilus galloprovincialis*), and Turolla et al. (2020) (*Ruditapes philippinarum*).

Shellfish have been recently proposed as an alternative method to carbon (C) sequestration (Ray et al. 2018). Its potential and relevance has been debated (Filgueira et al. 2015, 2019; Ray et al. 2018) mainly for the lack of consensus on (1) which biological process needs to be taken into account, (2) how to include them, and (3) the relevant scale to apply the CO_2 budget (Alonso et al. 2021). According to Filgueira et al. (2015, 2019) the carbon accounting in shellfish is mainly based on the balance between (1) the CO_2 emitted by the respiration, (2) the C fixed in the shell, and (3) CO_2 release by the biogenetic precipitation of calcium carbonate (CaCO_3) during the formation of the shell, while according to the same author, the (4) food ingestion, (5) the biodeposition (feces and pseudo-feces), and (6) the fate of the shell are not generally considered (Filgueira et al. 2015, 2019; Ray et al. 2018).

LCA studies that include carbon balances take into account carbon fixation in the shell of the mussel and the CO_2 emitted during its formation. However, the CO_2 emitted during respiration is not counted as it is assumed that CO_2 from respiration enters a short biological cycle that recycles CO_2 from respiration through autotrophic processes (photosynthesis) (Ray et al. 2018). This procedure is applied in Aubin et al. (2018) and Fry (2012). Furthermore, the CO_2

release during the formation of shell is accounted only by Turolla et al. (2020). According to Ray et al. (2018), the emissions produced by the precipitation of CaCO_3 do not belong to the biological cycle and thus should be taken into account. The authors estimate that this increases the CC impact category up to 250% (Ray et al. 2018). Finally, Lourguioui et al. (2017) include food ingestion, biodeposition, respiration, and carbon fixation in the shell into the C balance by applying a model proposed by Brigolin (2009) to estimate the biomass yield and nutrient fluxes of a mussel farm.

Alternatives to include the flux of nutrients P, N, and C (e.g., associated with the biodeposition, respiration, and nutrient fixation in the shell) into LCAs can be found in the literature. Baldan et al. (2018) provide an R package (*RAC*) to account for the C, N, and P content in feces and pseudofeces for different shellfishes. The *RAC* package is built upon the formulas proposed by Brigolin (2009) for the Mediterranean mussel (*Mytilus galloprovincialis*) and Solidoro et al. (2009) for the Manila clam (*Ruditapes philippinarum*). Moreover, Ray et al. (2018) propose a simple formula based on (1) the shell mass at harvest, (2) the percentage of shell made up of CaCO_3 , and (3) the ratio of CO_2 released per CaCO_3 to account for the emissions released by the shell formation. It is worth mentioning that as (3) depends on the pH, temperature, and salinity of the area the author used the R package (*seacarb*) to easily estimate the ratio. Such developments may help to have a better understanding of the potential benefit of shellfish to sequester carbon.

While the inclusion of positive benefits from the uptake of nutrients and the sequestration of C into the studies may add complexity and be target of discussion, the results provide a more accurate picture of the reality and therefore should be encouraged. Studies accounting for such benefits should be supported with a sensitivity analysis and results should be provided with and without the potential benefit in order to ensure transparency and facilitate comparisons with other studies as done in Aubin et al. (2018) and Fry (2012).

Not covered by any of the reviewed documents is the natural condition of the farming area. Shellfish are mostly cultured in open aquaculture systems (OAS) and their yields and potential environmental impacts depend to a large extent on physical, chemical, and biological factors such as temperature, salinity, phytoplankton content in the water, sediment fluxes, oxygen fluxes, currents, and tidal conditions rather than on cultivation technique (Pahri et al. 2015). For example, the growth of the blue mussel in the Baltic Sea was found to be positively correlated with the salinity and the phytoplankton content in water in Gren et al. (2009); additionally, temperatures in the range of 8.3 °C and 20 °C are suggested as optimal for the growth of the blue mussel (Kittner and Riisgård 2005). Moreover, the eutrophication impacts of OAS rely on the natural

conditions of the waterbody, which are also influenced by external natural and anthropogenic factors due to the absence of physical boundaries (Pahri et al. 2015). Furthermore, as mentioned above, factors such as the pH, temperature, and salinity determinate the CO_2 released during the formation of the shell, thus significantly influencing the CC impact category.

Knowing the natural state of the waterbody and the seabed will help to better understand whether the potential environmental burdens are produced by the system itself or by the natural conditions of the area (Pahri et al. 2015). For doing so, data can be collected through observations, secondary data, or fieldwork. Statistical analysis, whenever necessary, may be used to measure whether there is any significant variation on the studied variables during the farming period (Pahri et al. 2015). For detailed review of the influence of natural factors on aquaculture systems, see Pahri et al. (2015). The collection of such data, however, may be limited by constraints in time and budget. Nonetheless, data of such physiochemical factors are, in many cases, available free of charge in different databases, e.g., the European Union's Earth Observation Programme (Copernicus) (CMEMS 2021). Such data can be used, due to the lack of LCA methods, in parallel to the life cycle inventory assessment (LCIA) to determine whether the farm alters the natural local conditions potentially affecting the ecosystem (see Sect. 4.5 for more detail how it may be implemented).

Regarding the technology, shellfish are cultured worldwide in a wide range of techniques. Mussels are produced in on-bottom techniques (i.e., beds), rafts, bouchots, longlines, and nets, while the longlines are the most worldwide spread technique (Lourguioui et al. 2017). Rafts and bouchots are considered regional practices mainly used in Spain and France, respectively (Aubin et al. 2018; Iribarren 2010). Although shellfish yields depend mainly on natural conditions of the area, the farming technique constrains to some extent the stocking density, e.g., around 80 tonnes/raft (Iribarren 2010) and 60 kg/bouchot (Aubin et al. 2018). Moreover, the material requirements and maintenance vary significantly across techniques; e.g., techniques different to the bouchot often use concrete for anchoring blocks (Aubin et al. 2018; Iribarren et al. 2010c; Lourguioui et al. 2017; Tamburini et al. 2020), the rafts and bouchots are made of wood, while the longlines are mainly based on nylon, polyethylene, and polypropylene (Tamburini et al. 2020). The life span of the materials also varies across materials, e.g., raft (20 years) (Iribarren 2010), bouchot (at least 15 years) (Aubin et al. 2018), and longlines (between 35 and 50 years) (Tamburini et al. 2020). The different material requirements for the techniques may significantly change the final environmental burdens of the farms. Transparent documentation helps practitioners to draw more detailed conclusions and to conduct fair comparisons.

4.3 Data and data quality

Reviewed documentation suggests that the type and quality of the data varies significantly across studies. Concerning the foreground system, primary data sources, e.g., interviews, questionnaires, and personal communication, are the most common sources of information. Additionally, data based on personal experiences of the authors have been used, e.g., in Tamburini et al. (2019, 2020). In general, inventory details of the data collected remain unpublished in most of the cases. Therefore, it is not possible to reproduce the results. This issue has been acknowledged in previous studies for aquaculture systems (Bohnes et al. 2019; Henriksson et al. 2012) and in LCAs in general (Ayres 1995; De Smet and Stalmans 1996). In such regards, only three studies provide, to some extent, information regarding the data and source used (Alvarenga et al. 2012; Spångberg et al. 2013; Tamburini et al. 2019). The latter provides complete information of the quantities and processes used to build the background system of the study.

Data for the studies ranged from single farm (Lourguioui et al. 2017) to samples of different farmers located in a specific area (Aubin et al. 2018; Iribarren et al. 2010c; Tamburini et al. 2019, 2020; Turolla et al. 2020). According to Henriksson et al. (2012) the foreground data quality in aquaculture depends on the region and the intensity of the system. Thus, detailed records of inventory may be better recorded in systems with high yields or in areas regulated by local or regional authorities. While Iribarren et al. (2010c) study an area with 98% of Spain's mussel production, the region studied by Aubin et al. (2018) is regulated by the "Appellation d'Origine Contrôlée" (AOC) label, and in Turolla et al. (2020), the production of clams is managed by cooperatives under the control of local authorities. Collecting data on smaller or artisanal producers as well from cooperatives, e.g., shared facilities (for depuration and packing), may (1) help to improve the data and their quality, (2) help to better understand the main stakeholders and incentives of the shellfish industry, and (3) provide evidence to strive sound political decisions.

Only one study by Tamburini et al. (2020) provides information on the representativeness of the data in a systematic manner. The authors use the pedigree matrix (Ciroth et al. 2016; Muller et al. 2016; Weidema and Wesnæs 1996), a semi-quantitative approach based on five different components, e.g., reliability, completeness, temporal correlation, geographical correlation, and a further technological correlation to account for the quality of the data. The main advantage of the pedigree matrix is that it has been introduced in the ecoinvent database and it can be used to conduct uncertainty analysis by means of Monte Carlo analysis simulation (Muller et al. 2016).

Data and data quality reporting is a concern in any LCA study. According to ISO 14044 (2006) data used for both foreground and background system should be clearly declared in order to meet the goal and scope of the studies. Thus, data sources as well as the treatment of missing data or assumptions should be transparently documented. The ISO standard, however, does not demand publication of all data used (Henriksson et al. 2012). Data published along with the studies are important to ensure transparency and reproducibility of the results. Thus, it is recommended to the extent of the possibilities to provide the data used to build the background without revealing sensitive information, e.g., the name of the suppliers. Regarding the foreground system, it is suggested to provide information on the material and processes used. Henriksson et al. (2012) suggest providing ecoinvent data ID numbers rather than the names of the ecoinvent data sets to create a comparable baseline across multiple studies. Moreover, it is advised to use pedigree matrices to systematically report the quality of the data used. This information will help practitioners in better understanding the models.

4.4 Allocation methods

Mussel shells can be treated as a co-product used to produce fertilizers (Spångberg et al. 2013), CaCO_3 (Alvarenga et al. 2012; Iribarren et al. 2010a), or as a substitute to sand used in concrete to fabricate artificial reefs (Carral et al. 2018). Allocation in shellfish LCAs, however, is to some extent avoided by treating the two potential products (flesh and shell) as one single product (Alvarenga et al. 2012; Iribarren et al. (2010a, b, c, d, e) and by assuming that the product is marketed mainly as fresh product.

Substitution approaches have been used in studies that attempt to add value to the waste (shell and organic remains). Iribarren et al. (2010a) assume that 100 tonnes of mussel shell and 100 tonnes of organic remains may avoid the production of 65 tonnes of CaCO_3 , and 192 tonnes of boiled ham, respectively. Similarly, Alvarenga et al. (2012) use the shell to avoid the mining extraction of CaCO_3 . Finally, Spångberg et al. (2013) state that system expansion was used, but it is not clear whether credits were granted to the system. In contrast to aquaculture of fish where the feed inputs are co-products of other systems (Henriksson et al. 2012), shellfish do not need to be fed and allocation arising from the food production is not required.

Allocation methods are controversial in LCA studies mainly because they can bias the results depending on the system under study. Thus, it should be well justified and documented. Moreover, its sensitivity should be tested in order to improve the robustness of the results (ISO-14044 2006).

4.5 Environmental impact assessment method

CML is the major impact assessment method used in the reviewed documents presumably because it is a well-documented method that includes more than 1700 different flows and characterization factors (Guinée et al. 2002) and is according to Parker (2012) the most commonly applied method in seafood LCAs. Additionally, recent articles use updated methods such as ReCiPe (RIVM et al. 2016).

Regarding the environmental impacts categories, CC, A, and E are assessed in all of the reviewed studies with the exception of two studies that only study CC (Fry 2012; Ziegler et al. 2013). These impact categories are commonly considered in LCA of agricultural (Spångberg et al. 2013) and aquaculture (Henriksson et al. 2012) systems. POF, OD, and MET are the most commonly included impact categories besides the above mentioned. Other impact categories like energy use (3/13 studies) and water depletion (2/13) are poorly covered.

Although the ISO 14044 (2006) does not suggest a particular impact assessment method, the selection of any particular method should be well justified. All reviewed studies documented the method applied, but the selection criteria remain unclear. Only Aubin et al. (2018) and Lourguioui et al. (2017) declared that the impact categories were selected based on (1) the frequency of use in previous studies (Lourguioui et al. 2017) and (2) on guidelines in aquaculture LCAs (Aubin et al. 2018). From the reviewed studies it is possible to argue that the selection of the method is mainly driven by the possibility to compare the results with previous studies applying the same method. Moreover, effort should be made in using more updated impact assessment methods (e.g., ReCiPe, an updated version of CML (Guinée et al. 2002), and eco-indicator 99 (Goedkoop and Spriensma 2001)) that allows to calculate 18 different midpoint indicators and three end-point indicators (RIVM et al. 2016). Additionally, ILCD (International Reference Life Cycle Data System) recommendations developed by the Joint Research Centre (JRC) of the European Commission aimed to establish a standardized common LCA impact method approach (ILCD 2018) could be used. In any case the characterization factors should be the latest available versions (Henriksson et al. 2012).

Environmental impacts associated with biodiversity are not discussed in any of the reviewed studies. Positive impacts are mainly associated with the provision of habitat by other organisms (formation of natural and artificial reefs), cycling of nutrients and the creation of sediment, and increasing seabed roughness that potentially can lead to a richer ecological communities under the farm and the surroundings (van der Schatte Olivier et al. 2020). On the other hand, shellfish farming can cause alteration of

biodiversity mostly by biodeposition of (pseudo) feces. Biodeposition may increase local biogenic sedimentation, affects oxygen fluxes, and potentially change the faunal community structure of the area under and in the vicinity of the farm (Carlsson et al. 2010; Nizzoli et al. 2005; Ysebaert et al. 2009). Moreover, shellfish farming may potentially increase the risk of species invasion (introduction of non-native species) (Aubin et al. 2018) and the transfer of diseases (Brenner et al. 2014). Such impacts are rarely covered in LCA methodologies (Cao et al. 2013; Henriksson et al. 2012; Pahari et al. 2015).

The challenge of taking biodiversity impacts into account in LCA has been discussed for more than 20 years (Chaudhary et al. 2015; Curran et al. 2011; Winter et al. 2017). Most of the attempts to include biodiversity in LCA are based on impacts of land use change (Winter et al. 2017), which is considered as the main driver of global biodiversity loss (Chaudhary et al. 2015; Winter et al. 2017). Additionally, Winter et al. (2017) and Gabel et al. (2016) provide an updated review of available methods (e.g., for land use/occupation, water use/consumption, invasive species, overfishing) and indicators (e.g., ecological footprint, ocean health index, water quality index for biodiversity, freshwater quality) to assess biodiversity in LCAs. Both reviews conclude that, despite the availability of methods to include biodiversity impacts into LCA, they are rarely used in common LCA practice presumably because (1) there are limitations posed by the LCA methodological framework; i.e., impacts in LCA are required to be generic in space, be free of interaction between impact pathways, and be strongly linked to a functional unit (Curran et al. 2011); (2) the lack of operational models for assessing biodiversity; and (3) the lack of interest in the topic of the practitioners (Winter et al. 2017).

Additionally, due to the specific biological and physico-chemical conditions in water bodies, land-based impact categories may not reflect impacts on aquatic systems well. Thus, the development of aquaculture-specific impact categories (e.g., biotic resource depletion, seafloor disturbance) should be endorsed (Aubin 2013; Bohnes et al. 2019; Bohnes and Laurent 2019; Cao et al. 2013; Henriksson et al. 2012; Pahari et al. 2015).

One potential alternative to cope with the above mentioned limitations of including biodiversity and seafloor disturbance impacts into shellfish and aquaculture LCAs is to combine the LCA method with other assessment tools, as is recommended by Bohnes et al. (2019). For example, analysis of variance (ANOVA) was applied in studies such as Ysebaert et al. (2009), Danovaro et al. (2004), Moriki et al. (2008), Christensen et al. (2003), and da Costa and Nalesso (2006) to assess the interactions of shellfish farming with the environment (sediment and biodiversity assessment). The ecosystem services such as increasing water transparency,

promotion of macrophyte growth (Schröder et al. 2014), and the promotion of habitat (Ysebaert et al. 2009) can also be measured. However, it is worth mentioning that although statistics techniques are relatively simple to conduct, the data required (i.e., sampling collection) are intensive in time and resources and therefore their application may be limited.

4.6 Interpretation methods

While all the reviewed studies conduct a contribution analysis, only four studies (30% of the total sample) conduct a sensitivity analysis to test different scenarios and hypothesis and an uncertainty analysis to the foreground data to test the variability of the results (Aubin et al. 2018; Lourguioui et al. 2017; Tamburini et al. 2019, 2020). Sensitivity analysis has been used to test different scenarios, mainly associated with the consumption of energy and materials, uptake of nutrients, and EoL alternatives. Regarding the uncertainty analysis, the Monte Carlo simulation (1000 iterations) is the most common method applied (Muller et al. 2016). The Monte Carlo simulation is a widely used method to perform error propagation for model parameters (Lo et al. 2005).

The ISO standard states that the interpretation phase comprises the evaluation of the results based on the limitations and main assumptions made in conducting the study. Moreover, conclusions and recommendations should be provided (ISO-14044 2006). For doing so, a complete analysis of the results should be provided to test whether the main limitations or assumptions within the study have a significant effect on the results. Thus, it is suggested in agreement with Henriksson et al. (2012), Aubin (2013), Henriksson et al. (2015), and Bohnes and Laurent (2019) that efforts should be made to ensure the robustness of the results by means of a sensitivity and an uncertainty analysis with the support of statistical tools (e.g., Monte Carlo simulation).

5 Conclusions

This work conducts the first critical review of shellfish LCA studies. For doing so, a literature review was conducted initially with 83 documents from which only 13 satisfied the selection criteria for the review. Results reveal that (1) shellfish LCAs are found for mussels, clams, and oysters, from which mussels, particularly *Mytilus galloprovincialis* and *Mytilus edulis*, are the most studied. A few studies assess the environmental impact of clam and oyster farming; (2) geographically, studies concentrated in Europe (mostly Spain and Italy) make up the majority of the studies while, Brazil, Algeria, France, Norway, and Sweden are represented by one study each.

The following methodological trends were identified: (1) most of the studies conduct an analysis of the farming and the packaging stages. Full LCAs from cradle to grave

are needed to better understand the environmental impact associated with the shellfish supply chain; (2) allocations are avoided in the studies since the functional unit is usually defined as whole fresh shellfish comprising of shell and flesh as it is marketed; (3) CML method is the most applied environmental impact assessment method, and the impact categories of CC, A, and E are the most commonly assessed; (4) scenario analysis is used frequently to test assumptions associated with the consumption of energy and materials, uptake of nutrients, and EoL alternatives; and (5) capital goods linked to the post-farming stage are mainly omitted from the system, mainly due to the lack of reliable data and because it is assumed that in a long life span the environmental impacts associated to capital goods become negligible.

Positive benefits from the shellfish farming are included only in some studies whether for the uptake of nutrients during the growing cycle or for delivering additional services (included by means of the substitution method). Credits for the uptake of nutrients have been included in fewer studies (38% of the total sample), most of them associated with nitrogen and phosphorous. Credits for the removal of carbon were only granted in three studies owing to the fact that the carbon sequestration potential of shellfish farming is still controversial (Ray et al. 2018). While the discussion whether shellfish farming may be considered an alternative to sequester C is open, the uptake of nutrients (nitrogen and phosphorous) has been well documented and therefore it is suggested to consider such benefits in future studies. Additionally, benefits have been granted in some studies for avoiding the conventional mining extraction of CaCO_3 and the production of mussel pâté.

Effort should be made in (1) moving forward to most updated assessment methods such as the ILCD or ReCiPe method as well with the inclusion of more environmental categories; (2) applying sensitivity and uncertainty analysis to test the significance of the results and provide better support to the discussions and recommendations; (3) conducting additional assessments, when is possible, to account for the local and regional impacts on the water bodies and the vicinity (e.g., sediment and biodiversity assessment); (4) collecting data related with the capital goods involved in the systems; (5) providing data associated with the back and foreground system as online resources in order to facilitate comparison and replication of the results; and (6) provide information regarding the natural conditions of the farming area as they may determine not only the total yields but also environmental needs and impacts (e.g., eutrophication and climate change) of the shellfish farming.

Finally, future studies on shellfish production and its environmental impact should increase the resolution and explore other farmed species, regions, and culture techniques in order to fill the knowledge gap of shellfish aquaculture and advance to a sustainable shellfish aquaculture.

6 Data availability.

All data generated or analyzed during this study are included in this published article.

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