Aquacultural Engineering 92 (2021) 102130

Contents lists available at ScienceDirect

Aquacultural Engineering

journal homepage: www.elsevier.com/locate/aque

Review

Life cycle assessment of aquaculture systems: Does burden shifting occur with an increase in production intensity?

Ramin Ghamkhar[a,1], Suzanne E. Boxman[b,1], Kevan L. Main[c], Qiong Zhang[b], Maya A. Trotz[b], Andrea Hicks[a,*]

[a] Department of Civil and Environmental Engineering, University of Wisconsin-Madison, 1513 University Ave, Madison, WI 53706, USA
[b] Department of Civil and Environmental Engineering, University of South Florida, 4202 E Fowler Ave. ENB 118, Tampa, FL 33620, USA
[c] Directorate of Fisheries and Aquaculture, Mote Marine Laboratory, 1600 Ken Thompson Parkway, Sarasota, FL 34236, USA

ARTICLE INFO

Keywords:
Life cycle assessment
Intensive aquaculture
Extensive aquaculture
Environmental impacts
RAS

ABSTRACT

Life cycle assessment (LCA), a tool used to assess the environmental impacts of products and processes, has been used to evaluate a range of aquaculture systems. Eighteen LCA studies were reviewed which included assessments of recirculating aquaculture systems (RAS), flow-through systems, net cages, and pond systems. This review considered the potential to mitigate environmental burdens with a movement from extensive to intensive aquaculture systems. Due to the diversity in study results, specific processes (feed, energy, and infrastructure) and specific impact categories (land use, water use, and eutrophication potential) were analyzed in-depth. The comparative analysis indicated there was a possible shift from local to global impacts with a progression from extensive to intensive systems, if mitigation strategies were not performed. The shift was partially due to increased electricity requirements but also varied with electricity source. The impacts from infrastructure were less than 13 % of the environmental impact and considered negligible. For feed, the environmental impacts were typically more dependent on feed conversion ratio (FCR) than the type of system. Feed also contributed to over 50 % of the impacts on land use, second only to energy carriers. The analysis of water use indicated intensive recirculating systems efficiently reduce water use as compared to extensive systems; however, at present, studies have only considered direct water use and future work is required that incorporates indirect and consumptive water use. Alternative aquaculture systems that can improve the total nutrient uptake and production yield per material and energy based input, thereby reducing the overall emissions per unit of feed, should be further investigated to optimize the overall of aquaculture systems, considering both global and local environmental impacts. While LCA can be a valuable tool to evaluate trade-offs in system designs, the results are often location and species specific. Therefore, it is critical to consider both of these criteria in conjunction with LCA results when developing aquaculture systems.

1. Introduction

Finfish and other aquatic animals are critical to providing a high-value protein source and important micronutrients for much of the world. As production from capture fisheries remains stable (FAO, 2018), aquaculture's critical role in meeting increased demand for aquatic food products is driving researchers to assess the sustainability of the industry. In addition, consumers are becoming increasingly concerned with the environmental and ethical impacts of their food choices (Andersson, 2000). Considering aquaculture's major contribution to global food supplies and security, it is important to evaluate the current environmental impacts associated with aquaculture.

Life cycle assessment (LCA) is a tool used to quantify local, and global environmental impacts of systems and processes. It is considered a "cradle to grave" analysis, meaning that the assessment includes raw material extraction through the final disposal of all components (Curran, 2006). LCA has become a valuable tool used to evaluate a variety of systems, including biofuel production, wastewater treatment systems, agriculture, and aquaculture (Stokes and Horvath, 2006; De Vries and de Boer, 2010; Campbell et al., 2011).

* Corresponding author.
E-mail address: hicks5@wisc.edu (A. Hicks).
[1] These authors contributed equally to this work.

https://doi.org/10.1016/j.aquaeng.2020.102130
Received 4 August 2020; Received in revised form 23 October 2020; Accepted 26 October 2020
Available online 5 November 2020
0144-8609/

Prior LCA studies have looked at environmental impacts from fishing vessels and fleets, fish feed, and aquaculture systems. Avadí and Fréon (2013) reviewed 16 papers on LCAs of capture fisheries production. The review focused on differences in methodologies used to complete the LCAs. Reviews by Henriksson et al. (2012) and Bohnes et al. (2018) focused on differences in aquaculture LCA methodologies and looked at different types of aquaculture production systems from 12 and 65 studies, respectively. They found variability in the methodologies used and allocations made, and suggested that their needs to be a standardization of methodology and aquaculture specific impact categories. Variations in reporting methodological and data choices hinder direct comparison of different studies; however, important industry trends can still be seen by reviewing different LCA analyses of aquaculture.

Aquaculture systems vary in design and can be divided into two general categories linked to intensity of practice: extensive and intensive (Fig. 1). Intensive aquaculture systems, such as recirculating aquaculture systems (RAS), in which 90–99 % of system water is recycled (Badiola et al., 2012), are commonly cited as a more sustainable option for aquaculture production due to localized reduction in water inputs and nutrient discharges. However, the high energy and material requirements for RAS, which can contribute to greater global impacts, such as global warming potential, are not usually included when discussing the sustainability of intensive systems. Alternatively, extensive systems often require fewer feed and energy inputs (Naylor et al., 2000; Wirza and Nazir, 2020). Extensive systems potentially have fewer global environmental impacts, although the open system boundaries can result in greater direct ecological impacts, such as degradation of water quality (Stickney, 1994).

Due to potential variation in environmental impact associated with aquaculture production methods, this review compares high input intensive systems to low input extensive systems. The aim of this review was to comparatively evaluate studies on intensive and extensive aquaculture systems, within a LCA framework, to develop a more complete picture of the environmental trade-offs incurred due to intensification of aquaculture systems.

2. Materials and methods

Studies on aquaculture production systems were reviewed to compare differences in environmental impact. Papers were identified using web searches in the online database ScienceDirect and the internet search engine Google Scholar using combinations of the keywords: life cycle assessment, environmental impact, fisheries, aquaculture, recirculating aquaculture systems, and integrated aquaculture systems. Eighteen papers, that contained information on the pertinent aquaculture systems, were selected and discussed based on the following criteria: (1) Capture fisheries were neglected given the published review by (Avadí and Fréon, 2013); (2) Life cycle impact assessment were the primary aim of the studies; and (3) information on the nature of production system (intensive vs extensive) were clearly highlighted. A tabulated list of studies and important characteristics of each study is provided in Table 1.

The ISO 14040 four step methodology (goal and scope, life cycle inventory, life cycle assessment, interpretation) was used as a framework to compare the aquaculture LCA studies (Temizel-Sekeryan et al., 2020). The review is focused on variation in environmental impact of different aquaculture systems; however, an analysis of the goal and scope, system boundary, functional unit, allocation, and impact assessment methods were necessary to establish a baseline and facilitate comparison of each study's results. Subsequently the selected papers were compared according to the system processes commonly considered within the system boundaries, which included feed, energy, and infrastructure. Similarly, the common impact categories of land use, water use, and eutrophication potential were selected for in-depth analysis.

3. Results

3.1. Goal and scope

The goal and scope definition is the first step of an LCA. It should provide a clear statement of the study's purpose. Development of the scope is often comprised of an explanation of the system boundaries, functional unit, the impact assessment methodology, impact categories, and allocation used in the study. This step determines what information is included or excluded in the LCA and facilitates or hinders comparisons between studies.

The organization of this information varied in the studies reviewed. Some studies included it all in one goal and scope section (Aubin et al., 2009; Ayer and Tyedmers, 2009; Phong et al., 2011), but most divided the goal and scope into additional sections (McGrath et al., 2015; Badiola et al., 2017). Only a few studies included a clearly expressed goal within the goal and scope definition (Jerbi et al., 2012; Samuel-Fitwi

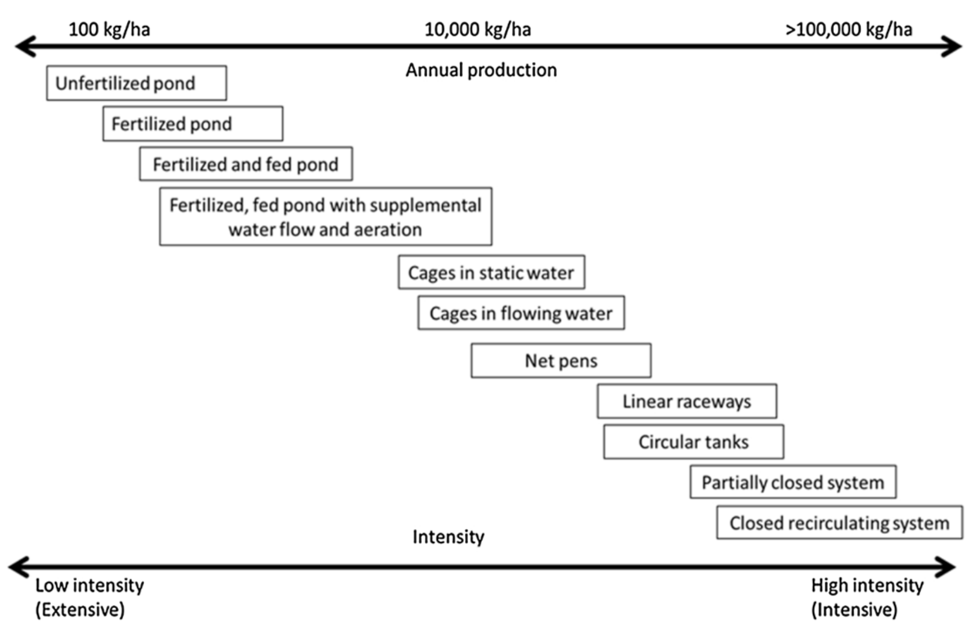

Fig. 1. Continuum of aquaculture production methods (adapted (Stickney, 1994)).

Table 1
List of studies included in literature review and important characteristics of each study.

Study	Systems included	Location	Species	Functional unit	Impact assessment method[1]	FCRs	Infrastructure included	Integrated with other animals/ plants
Aubin et al. (2009)	Flow through; sea cages; RAS	France, Greece	Rainbow trout; Sea-bass; Turbot	1 ton harvest ready live-weight fish	Papatryphon et al. (2004)	1.21, 1.77, 1.23	Yes	No
Aubin et al. (2006)	RAS	France	Turbot	1 ton live fish weight	Papatryphon et al. (2004)	1.23	Yes	No
Ayer and Tyedmers (2009)	Marine floating bag; land-based flow through; land-based RAS	Canada	Salmonids	1 ton harvest-ready live-weight fish	CML 2 Baseline 2000; CED v 1.03	N/R*	Yes	No
Efole Ewoukem et al. (2010)	Fish ponds integrated with pig manure, wheat bran, pig manure and crop by-products, or pig and chicken manure	Cameroon	Tilapia	1 ton fresh fish	CML 2 Baseline 2001; Aubin et al. (2009)	N/R*	Yes	Yes
Gronroos et al. (2006)	Net cage and land-based ponds	Finland	Rainbow trout	1 ton un-gutted rainbow trout after slaughtering	Individually calculated	1.255, 0.9, 1.53	No	No
Jerbi et al. (2012)	Traditional raceway, Cascade raceway	Tunisia	Sea bass, sea bream	1 ton live fish weight	CML 2 Baseline 2000; Papatryphon et al. (2004)	1.8, 2.1	Yes	No
Mungkung et al. (2013)	Net cage	Indonesia	Carp; tilapia	1 ton fresh fish to market	CML 2 Baseline 2000; CED v 1.03	1.7, 2.1	Yes	Yes
Pelletier and Tyedmers (2010)	Lake and pond	Indonesia	Tilapia	1 ton live-weight tilapia	CML 2 Baseline 2000; CED V1.03; Pelletier and Tydemers (2010)	1.7	No	No
Phong et al. (2011)	Fish ponds (high, medium, low intensity) integrated with rice fields or orchards	Vietnam	Fish	kilocalorie and kg per farm product	Individually calculated	N/R*	Not specified	Yes
d'Orbcastel et al. (2009)	Flow through; low head RAS	France	Trout (various sp.), artic char	1 ton of fish	CML 2 Baseline 2001	1.1, 0.8	Yes	No
Samuel-Fitwi et al. (2013)	Extensive flow through; Intensive flow through; RAS	Denmark, Germany	Rainbow trout	1 ton live trout	CML 2 Baseline 2000	N/R*	No	No
Wilfart et al. (2013)	RAS; semi-intensive pond; extensive polyculture pond	France	Salmon; common carp; tench; roach; perch; sander; pike	1 ton live fish	CML 2 Baseline 2001; CED v 1.05	0.95 (resirculating system), 1.29, (extensive) 0.86 (semi-extensive)	Yes	No
Badiola et al. (2017)	Pilot scale RAS	Northern Spain	Atlantic cod	1 kg grown-out cod, before slaughtering	CML 2 Baseline 2000	1.57	No	No
Abdou et al. (2017)	Circular net-cage (sea-cage)	Tunisia	Seabass & Seabream	1 ton of fish at the fish farm gate	CML 2 Baseline 2000	1.88 (seabass) 1.85 (seabream)	Yes	No
Biermann and Geist (2019)	Pond aquaculture	Southern Germany	Common carp	1 kg of live carp at the farm gate	International Reference Life Cycle Data System (ILCD)	2	No	No
McGrath et al. (2015)	Floating aquaculture (SWAS)	Canada	Chinook salmon	1 ton live-weight salmon	- ReCiPe 1.07 - CED 1.05 - Papatryphon et al. (2004)	1.459	Yes	No
Henriksson et al. (2017)	Conventional aquaculture + management practices applied	Egypt	Tilapia	1 ton live tilapia at farm gate (mass & economic allocation)	–5th IPCC assessment report - CML Baseline 2013	1.39-1.82	No	Yes
Yacout et al. (2016)	concrete ponds (intensive & semi-intensive)	Egypt	Tilapia	1 ton live tilapia at farm gate	Eco-Invent database 2008	NR*	No	No

* NR: Not Reported.

et al., 2013; Yacout et al., 2016; Abdou et al., 2017; Henriksson et al., 2017). Many included a goal in the introduction (Aubin et al., 2006; Gronroos et al., 2006; Aubin et al., 2009; Ayer and Tyedmers, 2009; d'Orbcastel et al., 2009; Efole Ewoukem et al., 2010; Phong et al., 2011; Wilfart et al., 2013; Biermann and Geist, 2019). In general, the goals of the reviewed studies were to quantify or evaluate the environmental impacts of the studied systems, while some included comparisons of different systems or operational scenarios (Biermann and Geist, 2019).

3.2. System boundaries

The system boundaries define what processes are included in the LCA. In its most basic form, this includes all processes from cradle to grave (Fig. 2, the system boundaries for cradle to farm-gate and cradle to grave are shown; the inclusion of dashed processes varies with study). System boundaries of food product studies often stop at farm-gate and do not include processing, retail, or household use (Henriksson et al., 2012). Most of the reviewed studies used a boundary of cradle to farm-gate (McGrath et al., 2015; Yacout et al., 2016; Abdou et al., 2017; Badiola et al., 2017; Henriksson et al., 2017; Biermann and Geist, 2019). Aubin et al. (2009) and Mungkung et al. (2013) only looked at hatchery to farm gate. While Gronroos et al. (2006) used a system boundary that ended at delivery to additional processing or retailers and included packaging materials, production, and manufacture.

Within the defined boundary, each aquaculture system was broken into different processes. The classification of these components is up to the author's discretion and varied among the papers reviewed. Aqua-feed, diet, or feed components were included in all studies. Energy carriers (e.g. electricity, natural gas, gasoline) or electricity production were also commonly reported as a separate process. If energy carriers were not included as a separate process they were included within other processes (Gronroos et al., 2006; Pelletier and Tyedmers, 2010). In the three studies where agriculture was integrated with aquaculture (Efole Ewoukem et al., 2010; Phong et al., 2011; Mungkung et al., 2013), energy was included in the system boundary but was not isolated as an individual process.

Across industries, infrastructure and capital goods have been excluded from LCAs based on the assumption that the impacts are relatively small (Frischknecht et al., 2007; Henriksson et al., 2012). Specifically within aquaculture, Ayer and Tyedmers (2009) reported that infrastructure's impacts were negligible in salmon production. Based on the results of Ayer and Tyedmers (2009), studies by Pelletier et al. (2009); Pelletier and Tyedmers (2010) excluded infrastructure in their LCAs. The studies that were more likely to include infrastructure as a process were those that evaluated either land-based RAS or flow-through systems. Samuel-Fitwi et al. (2013) looked at RAS and flow-through systems, but provided no justification for excluding infrastructure in the LCA. Most studies that looked at ponds or net cages did not include infrastructure except Efole Ewoukem et al. (2010).

3.3. Functional unit

LCA relates the environmental impact to the production system through the functional unit (FU). The FU quantifies the intended purpose of the production system. Comparisons between different systems are only possible if they have the same FU. Typically the FU is based on the primary product produced but can be refined to include temporal and quality criteria for a more complete description of the system function (Cooper, 2003; Avadí and Fréon, 2013).

The papers reviewed used similar FUs, in that they were mass quantities of fish. The amount of post-harvest processing, species, and quantity varied between papers. In general, all the FUs were variations on either 1 kg or 1 ton live-weight fish. Phong et al. (2011) studied an integrated agriculture-aquaculture system with multiple products and therefore used two FUs: kilocalorie and kg individual farm product.

3.4. Allocation

Many systems have multiple products, which poses a challenge when estimating the environmental impact. The environmental impact is not necessarily equally divided among the multiple outputs or co-products. Material and energy flows attributed to co-products must be allocated in a systematic way (Henriksson et al., 2012). The ISO-Norm (2006) describes a three step hierarchy to address allocation issues: 1) avoid allocation through subdivision or system expansion, 2) use allocation based physical relationships, 3) use allocation based on another non-physical relationship.

Five papers used economic allocation to divide environmental impacts among co-products where necessary. In Ayer and Tyedmers (2009) and Pelletier and Tyedmers (2010), the gross nutritional energy content was used to allocate environmental burdens. Allocation by gross nutritional energy content has been proposed as appropriate for seafood production because it incorporates the main function of aquaculture, chemical energy production in the form of food (Ayer, 2007). Ayer and Tyedmers (2009) also used system expansion to account for recovered fish waste in a RAS. To account for the use of fish waste as an organic fertilizer, an offset of an equivalent amount of chemical fertilizer was applied. In Gronroos et al. (2006), allocation was avoided by using whole fish as the functional unit to prevent allocation issues with co-products during processing.

3.5. Impact assessment methods

Life cycle impact assessment involves selecting impact categories and assigning associated characterization factors to the materials and energy inputs and outputs (Avadí and Fréon, 2013; Wu et al., 2020). A standardized method is often used to apply the characterization factors

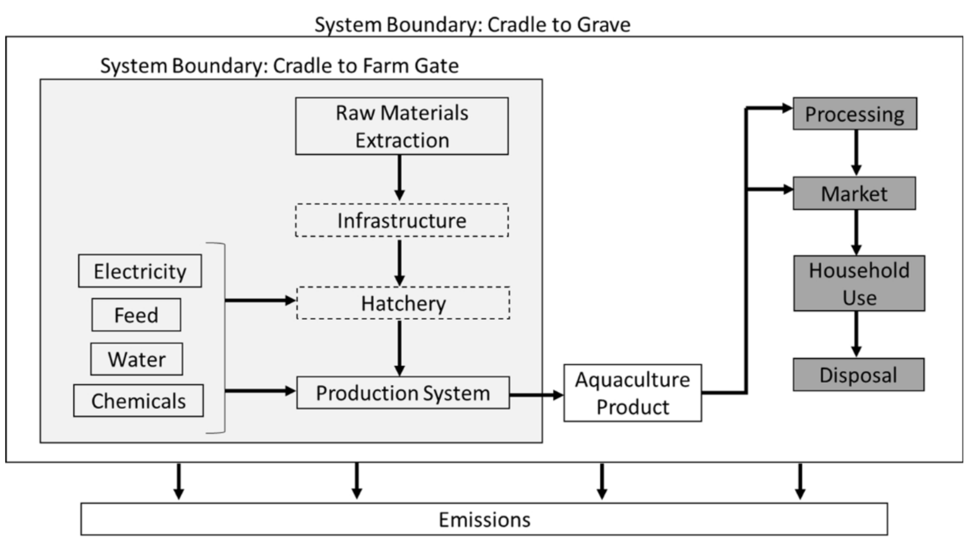

Fig. 2. Generalized system diagram for aquaculture systems.

to the life cycle inventory results; however, some methods are calculated independently (Avadí and Fréon, 2013). A wide range of impact categories and characterization methods have been used for aquaculture studies. The dissimilarity of impact categories used can impede comparison between studies, similar to difficulties with different system boundaries or functional units.

In total, twenty three different impact categories were used (Table 2). The CML baseline method was the only standardized method used to calculate common impact categories, such as eutrophication potential, acidification potential, and global warming potential. Studies that did not use the CML baseline method or had additional impact categories, used independent methods for characterization.

All studies included eutrophication and acidification potentials. For the characterization of eutrophication potential, Gronroos et al. (2006) individually considered eutrophication of aquatic and terrestrial systems, and used characterization factors specific to Finland for each distinguished impact factor (as opposed to using standardized eutrophication impact characterization and assessment methods). A measure of kg CO_2 equivalents was included in all the studies termed either greenhouse gas emissions or climate change. Energy use was considered in all but two of the investigated papers; five different terms were used and three different units.

The above impact categories are all measures of abiotic (non-living) resource use; however, in food production, biotic (living) resources are also consumed. Net primary production (NPP) can be used as a quantifiable measure of biotic resource use. The calculation of NPP use (NPPU) is based on the principle that plants convert sunlight into chemical energy and store it as carbon complexes. These carbon complexes move between trophic levels losing efficiency as carbon is transferred to higher trophic levels. NPP is a finite resource, using it as an impact category can help identify areas of inefficient resource allocation and can be used to improve the ecological efficiency of aquaculture (Pelletier and Tyedmers, 2007). NPPU measured as kg C was used as a characterization factor in eight of the papers reviewed. Most papers used the methodology described in (Papatryphon et al., 2004). Only Pelletier and Tyedmers (2010) calculated biotic resource use with methods described in Pauly and Christensen (1995) (narrowly described later by Ghamkhar and Hicks (2020)).

In seven of the reviewed papers, land or surface use was used as an impact category. Land use encompasses the alteration of land directly through the removal of natural landscape due to deforestation, agricultural practices, or construction of impervious surfaces (Brentrup et al., 2002). The assumption is that land should be conserved and excessive loss of land due to human development, has negative impacts on the environment (Brentrup et al., 2002). Land use or land use occupation is typically measured as an area over time, annual cubic meters (m^2a) or cubic meters per year (m^2yr^{-1}) (Mattila et al., 2011). Each paper independently calculated land use and accounted for surface area occupied by crops for feed production and area occupied by physical aquaculture systems in m^2, m^2a, or m^2/year.

Land use is one method to connect natural resources with aquaculture. Water use or water dependence are also measures of natural resource depletion. In aquaculture, water use is of particular importance because some production systems, like flow-through systems, are criticized for high volumes of water use, while others like RAS are commended for low water use. Incorporating this impact category, can provide information about possible burden shifting of decreased water use. Nine of the reviewed studies incorporated water use/water dependence as an impact category measuring m^3 of water flowing into production systems.

3.6. Impact assessment results and interpretation

Interpreting the results from the impact assessment is the final step of a LCA. The purpose of the interpretation step is to translate the results from the impact assessment into general conclusions about the type of environmental impact (global warming, eutrophication, etc.) and the system processes that contributed greatest (feed, energy, etc.). In the sections below, the results from three processes (feed, energy, and infrastructure) and three impact categories (land use, water use, and eutrophication) are discussed in depth.

3.6.1. Feeds

In the reviewed papers that compared different types of aquaculture systems in relation to feed, feed typically had the greatest environmental impact on NPPU and energy use (EU). In d'Orbcastel et al. (2009), a comparison between a RAS and a flow-through system for the production of trout showed that feed contributed greatest to NPPU (21,432 to 28,126 kg C) and energy (17,746 to 23,289 MJ). A sensitivity analysis on the feed conversion ratio (FCR) showed a reduction in NPPU and energy use could be achieved if the FCR of the RAS was decreased from 1.1 to 0.8. While the suggested 0.8 FCR was based on an experimental RAS, this level of efficiency is achievable in RAS producing various trout (d'Orbcastel et al., 2009; Buric et al., 2014), and salmon (Carter et al., 2003; Davidson et al., 2016; Ghamkhar and Hicks, 2020) species.

Similar results from a reduction in FCR were found in Jerbi et al. (2012) comparing two types of flow-through systems. Feed contributed approximately 40,000 kg C, which could be due to the higher FCRs of 1.89 and 2.11. Estimates of energy use from feed for the systems of Jerbi et al. (2012) ranged from 29,000 MJ to 33,412 MJ (system total energy use ranged from 170,000–280,000 MJ), and these were also likely higher than in d'Orbcastel et al. (2009) due to the higher FCRs. Aubin et al. (2009) compared a trout flow-through system (FCR = 1.21), sea-bass cages (FCR = 1.77), and a turbot RAS (FCR = 1.23). Similar as above, feed production contributed greatest to NPPU and EU. The NPPU was 62,200, 71,400, and 60,900 kg C for the flow-through, cage, and RAS respectively. The values are similar to those found in Jerbi et al. (2012), but greater than those found in d'Orbcastel et al. (2009) possibly due to the variations in system boundaries despite similar FCRs.

Abdou et al. (2017) compared the environmental impacts of seabass (FCR = 1.88) and seabream (FCR = 1.85) production in a sea-cage aquaculture farm. Regardless of the system output, feed production posed the greatest impact contribution to NPPU (>99 %) and cumulative energy demand (71–79 %).

McGrath et al. (2015) evaluated the environmental impacts of salmon production (FCR = 1.459), using a novel closed-containment aquaculture technology. In consistency with other studies, feed production contributed the greatest to NPPU (aka Biotic Resource Use - BRU, unit mass of C eq, 100 %). However, the contribution of feed production to energy use was slightly lower than on-site energy use (39.7 % vs. 42.1 %). The NPPU was 1429 Mg C and the cumulative energy use was 36,324 MJ for feed production. Fig. 3 provides a conceptual illustration of the general correlation among FCRs versus NPPUs and EUs associated with food production using a regression analysis. Despite the fact that the plotted trends pose a relatively low regression (R^2) due to allocation differences across studies as well as limited data availability, there was general upward trend in associated environmental impacts with an increase in FCRs.

The environmental impacts of feed can also change with intensity. In Samuel-Fitwi et al. (2013), three different system intensities were explored (extensive flow-through, intensive flow-through, and intensive RAS, EU were not incorporated). Impacts from feed decreased with increasing intensity for all investigated impact factors due to improved FCRs. Yacout et al. (2016) compared the environmental impacts of tilapia production in intensive and semi-intensive farms. Their assessment revealed that the production of tilapia in intensive farming has less impacts in global warming, acidification, and cumulative energy demand (despite higher impacts in eutrophication for intensive animal production, BRU and BRU categories were not incorporated).

As intensity increases, FCRs typically improve (Hasan and Soto, 2017) (Fig. 3), which results in overall decreased environmental impact, as shown with the sensitivity analysis on FCRs in d'Orbcastel et al.

Table 2
Impact categories used in reviewed LCA studies with reporting units.

Impact	AD*	GWP*	CC*	HTP*	MTP*	AP*	EP*	CED *	EU *	NREU*	TCED *	FEU *	NPPU *	LC*	LU*	SU*	LO*	WU/ WD*	Other **
Unit	kg Sb eq	kg CO_2 eq	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq	kg SO_2 eq	kg PO_4 eq	MJ	MJ	GJ and MJ	GJ	kJ	kg C	m^2a or m^2yr^{-1}	m^2/yr	m^2	m^2 /yr	m^3	N/A
(Aubin et al., 2009)			x			x	x		x				x					x	
(Aubin et al., 2006)		x				x	x			x (MJ)			x						
(Ayer and Tyedmers, 2009)	x	x		x	x	x	x	x											
(Efole Ewoukem et al., 2010)						x	x			x (GJ)			x		x			x	
(Gronroos et al., 2006)			x			x	x												
(Jerbi et al., 2012)		x				x	x		x				x			x (m^2/yr)		x	
(Mungkung et al., 2013)			x			x	x		x				x				x	x	
(Pelletier and Tyedmers, 2010)		x				x	x	x					x						
(Phong et al., 2011)			x			x	x (NO_3 eq)					x			x (m^2)				
(d’Orbcastel et al., 2009)		x				x	x		x				x			x		x	
(Samuel-Fitwi et al., 2013)		x				x	x							x				x	
(Wilfart et al., 2013)			x			x	x				x		x	x				x	
(Badiola et al., 2017)	x	x				x	x		x										
(Abdou et al., 2017)		x				x	x				x		x				x		x
(Biermann and Geist, 2019)			x	x		x	x												x
(McGrath et al., 2015)		x				x	x	x					x						
(Henriksson et al., 2017)		x				x	x		x						x			x	
(Yacout et al., 2016)		x				x	x	x										x	
Sum	2	11	6	2	1	18	18	4	6	2	2	1	10	2	3	2	2	9	2

* AD: Abiotic Depletion; GWP: Global Warming Potential; CC: Climate Change; HTP: Human Toxicity Potential; MTP: Marine Toxicity Potential; AP: Acidification Potential; EP: Eutrophication Potential; CED: Cumulative Energy Demand; EU: Energy Use; NREU: Non Renewable Energy Use; TCED: Total Cumulative Energy Demand; FEU: Fossil Energy Use; NPPU: Net Primary Production Use; LC: Land Competition; LU: Land Use; SU: Surface Use; LO: Land Occupation; WU: Water Use; WD: Water Dependence.

** Other: Impact categories that are either ad-hoc basis or case-specific.

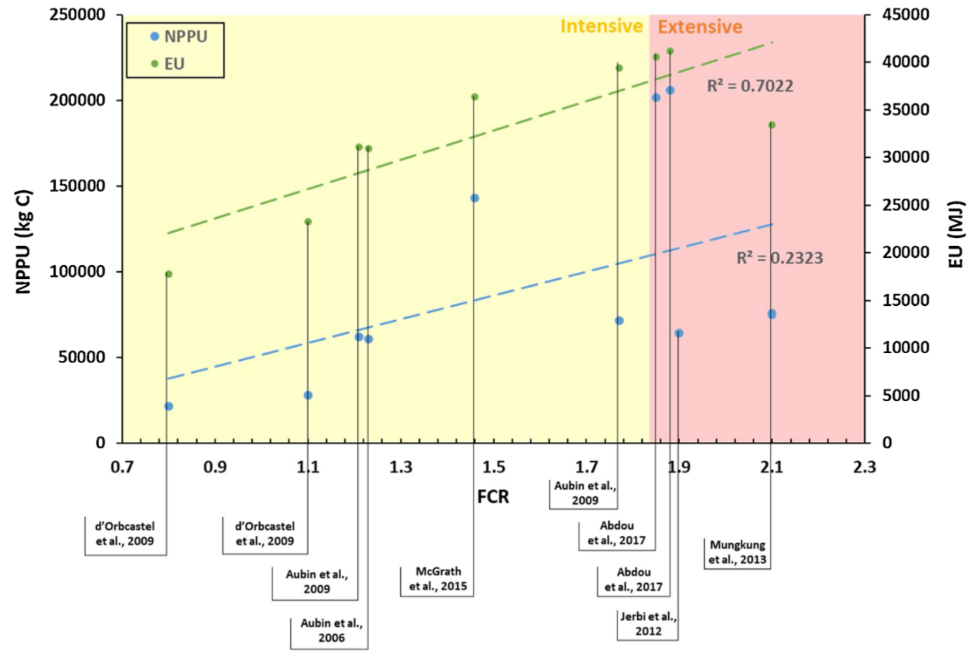

Fig. 3. Net Primary Production Use (NPPU, left vertical axis) and Energy use (EU, right vertical axis) among investigated studies with various reported feed conversion ratios (FCR, horizontal axis).

(2009). Mungkung et al. (2013) considered two net-cage systems with an intensive and semi-intensive stocking density. The systems were integrated such that they produced two species simultaneously. In the intensive, high density system the NPPU and energy use were 14,205 kg C and 28,645 MJ, respectively. These values were lower than in the semi-intensive, lower density system, which had an NPPU and energy use of 16,462 kg C and 32,945 MJ, respectively. Mungkung et al. (2013) concluded that the cause of this difference was due to the greater feed efficiency (i.e. lower FCR) in the intensive system.

In extensive systems, the relative contribution of feed is decreased because fertilizer, often in the form of animal manure, is added to increase primary production of algae and microorganisms on which the fish feed. Wilfart et al. (2013) looked at RAS and pond systems with two levels of intensity. The contribution of feed to NPPU was 333 kg C to 744 kg C because a lower quantity of the feed came from harvesting higher trophic level fishery resources.

Finally, it is important to mention that the implementation of FCR reduction strategies, regardless of feeding components, to mitigate the environmental impacts is not a coherent approach. Gronroos et al. (2006) looked at variations in feed, and found that improving the FCRs decreases the impacts of feed for all categories. However, this impact mitigation is attributed to changes in the feed composition (such as increasing the soy content). In Pelletier and Tyedmers (2010), crop and fisheries derived tilapia feeds were evaluated. The results from this assessment showed that the greatest contribution to NPPU was fish meal and fish oil used in pelleted feed. For example, fish oil uses over 40 times more kg C than palm oil. Cumulative energy demand was also greater from the fisheries derived components, however the margin was smaller. Fish oil was associated with 33,000 MJ and palm oil 4580 MJ.

3.6.2. *Energy*

Energy was used as a system process in several of the reviewed papers and was typically reported as either electricity or energy carriers. In papers that did not consider energy directly as a process, the impact category cumulative energy demand or energy use was used to draw conclusions about the aquaculture system's energy consumption and associated environmental impacts.

Intensive flow-through systems and RAS require large quantities of electricity for operation. When comparing flow-through systems and RAS, RAS typically have higher energy requirements due to the pumping requirements for water recirculation. In Ayer and Tyedmers (2009), electricity for the RAS had an energy demand of 291,000 MJ compared with a demand of 70,100 MJ for the flow-through system. The impacts of electricity are also seen in global warming potential. The RAS had a global warming potential of 23,700 kg CO_2 eq and the flow-through had a global warming potential of 1020 kg CO_2 eq associated with electricity. Other studies have found similar trends for energy in RAS and flow-through systems. Aubin et al. (2009) considered energy carriers as a process and compared three production systems, a cage system, flow-through system, and RAS. The energy use increased with higher on-farm energy consumption. The energy use for each system was 9191 MJ, 37,132 MJ, and 290,985 MJ for the cage, flow-through, and RAS, respectively. The global warming potential followed the same trend and was 163 kg CO_2 eq, 406 kg CO_2 eq, and 3670 kg CO_2 eq for the cage, flow-through, and RAS, respectively. The calculated global warming potential in Aubin et al. (2009) was low compared to the RAS in Ayer and Tyedmers (2009) despite similar energy use values because the latter evaluated system was located in France, where a higher proportion of electricity is produced by nuclear power plants. A sensitivity analysis in Ayer and Tyedmers (2009) illustrated the importance of the type of electricity generation. When the energy mix was varied to include less coal-based production and more hydroelectricity, the global warming potential decreased from 23,700 kg CO_2 eq to 10,300 kg CO_2 eq. In Badiola et al. (2017) two electricity scenarios (scenario 1: 100 % from non-renewable source, and scenario 2: 50 % from no-renewable – 50 % from renewable source) were evaluated to assess the environmental impacts of a RAS. In the scenario that 50 % of the electricity were provided from a renewable source (biogas from agricultural plants), global warming potential decreased from 21.64 to 14.74 kg CO_2 eq, based on average electricity consumption.

The source of electricity is not the only factor that impacts the energy process. In Wilfart et al. (2013), a turbot RAS required more energy (250,010 MJ), due to water heating and cooling requirements, than a salmon RAS (55,530 MJ). This highlights the importance of production type and specifications in energy consumption and associated impacts (Ghamkhar et al., 2020). The global warming potential followed the same trends. The turbot RAS had a global warming potential of 3670 kg CO_2 eq and the salmon RAS had a global warming potential of 417 kg CO_2 eq. A study comparing two flow-through systems also concluded that operational decisions influence environmental impacts (Jerbi et al., 2012). The flow-through systems with a cascade raceway had greater electricity use due to greater pumping requirements. The LCA results showed a higher total global warming potential of 17,500 kg CO_2 eq in the cascade raceway, with electricity contributing greatest to the global warming potential. d'Orbcastel et al. (2009) also evaluated different operational characteristics of aquaculture systems. When two different pumping scenarios were considered for flow-through systems, the high pumping scenario had a greater energy use and global warming potential.

Extensive systems have much lower energy requirements than the intensive systems discussed above. In Phong et al. (2011) electricity was included in the LCA, but not directly as a process. The contribution to impact categories was divided into on-farm and off-farm use. For the impact category of energy use, most of the use was attributed to off-farm activities, which includes inorganic fertilizer production, rice co-products, and feed. Since this study considered integrated agriculture and aquaculture, the authors also looked at the contribution of farm products to the impact categories. The on-farm energy use for pigs and fish were similar at 314 kJ/kg and 353 kJ/kg, respectively, and poultry was higher at 583 kJ/kg. Mungkung et al. (2013) looked at extensive pond systems that produced multiple fish products. Energy was not considered directly as a process, but the impact category of energy use was used. Similar to Phong et al. (2011), feed contributed most to energy use; the contribution of farm operation was negligible. Pelletier and Tyedmers (2010) considered the process of farm energy use for the pond and lake systems studied. The lake systems did not require aeration. As such, they had low energy use and less of the global warming potential was due to farm energy use. In contrast, the pond systems required more electricity for aeration and had higher energy use and global warming potential.

3.6.3. *Infrastructure*

In addition to energy, infrastructure is another factor that distinguishes intensive and extensive aquaculture systems. Intensive cage systems, flow-through systems, and RAS all have greater material requirements than extensive pond systems. In an LCA these material inputs are occasionally considered, but more frequently they are considered negligible and are excluded from the life cycle inventory (Boxman et al., 2017; Avadí and Fréon, 2013; Ghamkhar et al., 2020).

Ayer and Tyedmers (2009) included infrastructure and provided tables showing their inventory data. Of the four systems compared, the RAS and net-pen systems typically had high impacts from infrastructure. Most of the impacts from infrastructure were seen in the marine toxicity potential and the second greatest impact was to cumulative energy demand/energy use. Focusing on the marine toxicity potential and cumulative energy demand/energy use impact categories, the impacts from infrastructure were consistently much lower than the impacts of electricity or feed production. For example, in the RAS that had the highest impact to marine toxicity potential, infrastructure only contributed 0.13 %. In contrast, electricity production contributed 93 % of the marine toxicity potential.

Other studies that included infrastructure also reported that it contributed to less than 13 % of environmental impact for all impact categories included. Aubin et al. (2009) considered infrastructure impacts on three types of aquaculture systems. No trends were observed between production systems. The greatest impacts from infrastructure were to cumulative energy demand and climate change, but they were all less than 13 %. The other papers reviewed which considered infrastructure were d'Orbcastel et al. (2009); Jerbi et al. (2012); Mungkung et al. (2013); Wilfart et al. (2013); Abdou et al. (2017), and McGrath et al. (2015).

3.6.4. *Land use*

Land use (LU), land competition (LC), or surface use (SU) were impact categories considered in nine of the papers reviewed. Each term is associated with a different characterization method, since methods for inclusion of land use in LCAs are still debated (Mattila et al., 2011). Most of the papers reviewed used the method outlined in the Handbook on Life Cycle Assessment by Guinée (2002) developed by the Center for Environmental Studies, University of Leiden.

Collectively the results for the land use characterization factor, regardless of methodology or units used, indicated that feed production had the greatest impact on land use. Jerbi et al. (2012) investigated surface use measured in m^2/yr and found that the tank surface area occupied by a flow-through system was negligible when compared to the surface area associated with fish feed. d'Orbcastel et al. (2009) looked at surface use in m^2 and also found feed contributed more to surface than any other process. Feed contributed 2097 to 2736 m^2 of surface use, while other processes contributed 0.0–0.2 m^2. When FCR was decreased, the authors saw an associated decrease in surface use. At an FCR of 1.1, surface use from feed was 2752 m^2. When FCR was decreased to 0.8, surface use decreased to 2097 m^2. Two pumping scenarios, a high and a low scenario, were also considered in this study. The changes in pumping requirements did not impact surface area, further indicating the importance of feed to surface use. In Abdou et al. (2017), feed contributed to 98.7 % (1351.15 m^2/year) and 98.1 % (1311.38 m^2/year) of the land occupation per ton of seabream and seabass produced respectively.

A comparison of three different production system intensities in Samuel-Fitwi et al. (2013) found that electricity sources can also impact land competition. For the RAS studied in Samuel-Fitwi et al. (2013), feed contributed to 62 % of land competition, and electricity contributed to 38 % of land competition. When electricity generation was changed to include wind power in a sensitivity analysis, the total land competition dropped to 928 m^2a or about 37 % less. Due to the higher energy requirements, RAS had the greatest impact on land competition compared to both intensive and extensive flow-through systems. Moreover, due to the higher feed requirements in the extensive system, it posed higher land competition compared to the intensive system.

When compared to extensive systems, RAS had the lowest contribution to land competition in m^2yr, the extensive pond was second, and the semi-extensive pond was greatest (Wilfart et al., 2013). Instead of feed production, the on-farm fish production contributed to most of the land competition. Similar results were found in Efole Ewoukem et al. (2010), which compared the intensive flow-through system from Aubin et al. (2009) to several Cameroonian pond systems. The integrated pig and fish pond system (4369 m^2/year) had greater land use impacts than the flow-through system (2351 m^2/year). When compared to the other extensive pond systems in Cameroon, the impacts to land use decreased with decreasing productivity. The extensive systems studied in Phong et al. (2011) did not find land use significantly impacted by any of the processes included. When assessed on an m^2/kcal basis, all land use impacts were 0.023 m^2/kcal with no differences between on and off farm use.

3.6.5. *Water use*

Similar to land use, water use (WU) is a relatively new development in LCA characterization factors. It is important to consider in aquaculture production because one of the main benefits to developing RAS is the reduction in water use compared with extensive and semi-intensive production systems (Bohnes et al., 2018). In the papers reviewed, water use and water dependence (WD) was calculated based on direct water use, specifically the quantity of water flowing into the production systems. Mungkung et al. (2013); Henriksson et al. (2017) were exceptions and also indicated that the quantity of water used for crop irrigation was included in the water use. None of the papers reviewed considered indirect water use (the water used in the processing of underlying and upstream production chains, such as in feed production and electricity generation).

Aubin et al. (2009) found an increase in water use efficiency with increasing intensity. The RAS was the most water efficient, using 4.8 m^3, the cages used 52.6 m^3, and least efficient was the flow-through system, which used 48,782.2 m^3. Yacout et al. (2016) found similar results. Water use for intensive tilapia production was 200 m^3, while it was 35, 700 m^3 for semi-intensive tilapia production. When feed and pumping requirements were varied in d'Orbcastel et al. (2009), there was no change in the water use. A comparison of flow-through and RAS showed a 93 % reduction in water use. In Jerbi et al. (2012), the cascaded flow-through systems had a water dependence of 396,000 m^3 compared to only 190,000 m^3 in the traditional flow-through system. A comparison of two types of flow-through systems in Samuel-Fitwi et al. (2013), showed that the intensive flow-through system used only 1% of the water required in the extensive flow-through system. A RAS was also included in this comparison and it had 0% water use relative to the two flow-through systems.

In extensive systems, water use will vary with size of the ponds and production practices. The comparison of four pond systems in Cameroon showed that despite similarly sized ponds the water dependence varied and was not related to yield (Efole Ewoukem et al., 2010). The integrated pig and fish system had a water dependence of 16,900 m^3, whereas the pond fertilized with pig manure and crop by-products had a water dependence of 51,000 m^3 (i.e. 101.8 % increase). In Wilfart et al. (2013), water dependence was related to the pond surface area. The extensive pond in this study had the greatest water dependence of more than 41,000 m^3, the semi-extensive pond had a water dependence of 7500 m^3, and the RAS had a water dependence of 2500 m^3. Mungkung et al. (2013) and Henriksson et al. (2017) were the only authors to consider additional sources of water dependence. In Mungkung et al. (2013), irrigation for agriculture was included in particular water for rice production. When agricultural water dependence was considered, feed production contributed greatest to water dependence (71 %). High and low stocking density farming practices were considered. The low stocking density system had a higher water dependence of 1121 m^3 compared to 877 m^3 in the high stocking density system. In Henriksson et al. (2017), irrigation water on agricultural fields accounted for 7–12 % of the overall freshwater consumption, which represented the second largest consumer of freshwater.

The papers reviewed consistently show RAS to have lower direct water requirements and flow-through systems to have high water requirements. The extensive pond systems will vary with farming practices and pond age (Efole Ewoukem et al., 2010). Extensive pond systems can have water use similar to a flow-through system, while others might be more conservative and have lower water requirements. However, even under the conservative water use conditions, the impact will still be approximately 500 times greater than RAS.

3.6.6. *Eutrophication potential*

Eutrophication potential is based on nutrients, particularly nitrogen and phosphorous, emitted to environment. It is the one impact category that was included in all the papers reviewed. Like water use, the potential reduction in eutrophication potential is considered an advantage to RAS.

Several papers demonstrated lower eutrophication potential in RAS compared to flow-through or other production systems (Philis et al., 2019). Ayer and Tyedmers (2009), which compared four production systems, found RAS to have the lowest eutrophication potential. The resulting eutrophication was predominately attributed to feed and electricity processes. In the other systems, eutrophication was predominately due to grow-out emissions (production of juvenile to market size fish, as compared to smolt production, fuel use, and feed production). In the sensitivity analysis, changing the electricity mix to incorporate more renewables reduced the eutrophication potential of the RAS from 20.1 kg PO_4 eq to 11.6 kg PO_4 eq (i.e. by 42.3 %). Samuel-Fitwi et al. (2013) had similar results when comparing the extensive flow-through system, the intensive flow-through system, and the RAS, which had eutrophication potentials of 60.36 kg PO_4 eq, 60.03 kg PO_4 eq, and 4.04 kg PO_4 eq, respectively. In the flow-through systems, most of the eutrophication potential was due to fish production processes and in the RAS it was mainly due to electricity and feed processes. When the electricity was produced from wind power, the eutrophication potential for the RAS decreased by about half. Badiola et al. (2017) integrated energy audits in LCA of RASs with analogous results. In the scenario that 50 % of the electricity were provided from a renewable source (biogas from agricultural plants), eutrophication potential decreased from 0.04 to 0.02 kg PO_4 eq (i.e. 50 % decrease), based on average electricity consumption.

Reduced water discharges in RAS due to recirculation contribute to the lower eutrophication potential, but does not guarantee a RAS will have a low eutrophication potential. In Aubin et al. (2009), the differences between the flow-through system and RAS were reversed. The flow-through and RAS had eutrophication potentials of 66 kg PO_4 eq and 77 kg PO_4 eq, respectively. The higher eutrophication potential of the RAS was due to a higher protein content in the feed of 55 % compared to 45 % in the flow-through system. In d'Orbcastel et al. (2009) a flow-through system was also compared to a RAS. The eutrophication potential was reduced by 26–38 % in the RAS. The higher percent reduction was due to a lower FCR.

The eutrophication potential of a RAS will also vary depending on the facility. Wilfart et al. (2013) compared a RAS producing salmon and the turbot RAS studied in Aubin et al. (2009). The salmon producing RAS had a eutrophication potential of 34 kg PO_4 eq and the turbot RAS had an eutrophication potential of 77 kg PO_4 eq. The difference could be attributed to the higher energy use in the turbot facility, from heating and cooling the water. When the salmon RAS was compared to an extensive and semi-extensive pond system, the pond systems had lower eutrophication potentials than the RAS. The authors suggested that the lower emissions in the pond systems were due to internal nutrient cycling within the ponds which was not present in the RAS. Similar results reported by Yacout et al. (2016), comparing intensive and semi intensive tilapia production systems. Intensive production resulted in 14.1 kg PO_4 eq, while semi-intensive production resulted in 6.3 kg PO_4 eq (55 % less). The higher eutrophication impact in the intensive system were attributed to the intensive animal production in the aquaculture system.

In extensive systems, the eutrophication potential will depend on farm management practices. In Mungkung et al. (2013), the extensive pond and cage system that used feed more efficiently had a lower eutrophication potential. In Gronroos et al. (2006), the eutrophication potential was divided into aquatic and terrestrial based impacts and aquatic eutrophication was always greater than terrestrial eutrophication. In Biermann and Geist (2019), the eutrophication potential was divided in to terrestrial, freshwater, and marine eutrophication. Their evaluation, however, resulted in higher terrestrial eutrophication compared to marine eutrophication (for both conventional and organic carp production scenarios). While fish production generally contributes greatest to eutrophication potential, feed type also affects the overall emission-based environmental impacts. Decreasing the FCR can reduce the eutrophication potential as seen in Gronroos et al. (2006) and Mungkung et al. (2013). Over-fertilization of pond systems will also result in a high eutrophication potential (Efole Ewoukem et al., 2010). The eutrophication potential of the Cameroonian ponds ranged from 157 kg PO_4 eq to 908 kg PO_4 eq. These values are at least double the trout flow-through system, which had an eutrophication potential of 66 kg PO_4 eq. While pond systems have reductions in some global environmental impacts, locally they contribute to greater eutrophication potentials without the benefit of increased yields as in intensive systems.

3.6.7. *Monetary valuation of extensive vs. intensive production strategies*

To compare the overall favorability of increasing production intensity as a strategy to mitigate the overall environmental impacts, a trade-off analysis based on the pertinent impact categories and their relative impact level can be performed. In an effort to provide a comparative trade-off analysis of global and local impacts among intensive and extensive farming systems, GWP and EP have been selected, among the plethora of relevant indexes, as the most commonly investigated impact categories regarding global and local environmental impacts (Table 2)(Curran, 2006). Quantified GWP and EP impacts for the investigated studies who evaluated both impact categories (along with reported FCRs and cradle-to-gate system boundary), have been extracted from the literature and illustrated in Fig. 4.

An economic allocation of \$52/ton of CO_2 for GWP (social cost of carbon) and \$1.91/kg of PO_4 for EP have provided the opportunity to analyze and compare the impacts shift from extensive to intensive farming, considering an economic perspective. Social cost of carbon (for GWP) incorporates the long-term economic harms due to the net agricultural productivity changes, property damages from increased flood risk, human health, and changes in energy system cost (Council, 2013; Ghamkhar and Hicks, 2020). STEPWISE2006 (for EP) provides globally valid values to monetarise environmental impacts, using budget constraint method (Weidema, 2009; Pizzol et al., 2015). An adjustment of 1.13 (to convert EUR2003 to USD 2003) times 1.41 (to incorporate inflation from 2003 to 2020) has been executed.

As shown in Fig. 4, despite the fact that the plotted trends pose a relatively low regression (R^2) (due to allocation differences across studies as well as limited data availability), there is general downward trend in the associated overall monetary value of environmental impacts (GWP + EP) with a decrease in FCRs. This highlights the fact that shifting from extensive to intensive farming can be a potential approach to decrease the overall environmental impacts of aquafarming, considering both global and local indicators. However, it is important to mention that case-specific mitigation strategies (e.g. optimized heat in cold-weather setups, protein-rich feeds for carnivore species, etc.) needs to be investigated and applied to accomplish the overall improvement of farming environmental performance.

4. Discussion

In the first three steps of the LCA methodology, there are no specific patterns distinguishing intensive, semi-intensive, or extensive aquaculture production systems. The methodological choices are largely up to the author's discretion and intended goal. All the authors followed the guidelines developed by the International Standard Organization (ISO, 2008). The variation in functional units, system boundaries, allocation methods, and characterization factors does impede a direct comparison between LCA studies (Wu et al., 2019). As mentioned in Avadí and Fréon (2013) more standardization for fisheries practices would aid future LCA fisheries research. The analysis of specific processes and impact categories did reveal a tendency for increased intensity to result in a shift from local to global impacts for some environmental burdens (e.g. decrease in acidification potential and increase in global warming potential).

The impact of aquaculture feeds is well known to be one of the main impediments to development of sustainable aquaculture, which is further supported by this review (Basto-Silva et al., 2019; Bohnes and Laurent, 2019; Ghamkhar and Hicks, 2020). Both intensity level and FCR had clear impacts on the NPPU and cumulative energy demand/energy use of aquaculture systems (Gronroos et al., 2006; d'Orbcastel et al., 2009; Pelletier and Tyedmers, 2010; Mungkung et al., 2013). However, there are confounding effects to the impacts of feed between intensive and extensive systems. Extensive systems benefit from reduced feed requirements and therefore global environmental impacts due to supplemental primary production from fertilizers. The jump from extensive to intensive systems resulted in a large increase in global impacts from feed; however, more intensive systems can have also lower feed impacts due to improved efficiency and FCRs. Further improving FCRs is one way to reduce the impacts of feed. Although, at present, even with a low FCR, fish only incorporate 12%–25% of the nutrients from feed into biomass (Lucas et al., 2019). Alternatively, reducing the impacts from feed by improving the feed utilization of the whole system through production of a secondary species that used excess nutrients could increase the total system production and improve efficiency (Neori et al., 2004; Wu et al., 2019). These integrated multi-trophic aquaculture (IMTA) systems are suggested as a way to increase the environmental sustainability of RAS due to bio-mitigation of wastes and increase revenues (Barrington et al., 2009; Granada, 2015; Philis et al., 2019; Ghamkhar et al., 2020). The potential benefits of dual species production using the same amount of feed could also extend to reductions in electricity and fuel use due to greater production per unit of energy (Cederberg and Stadig, 2003; Bibbiani et al., 2018; Bohnes and

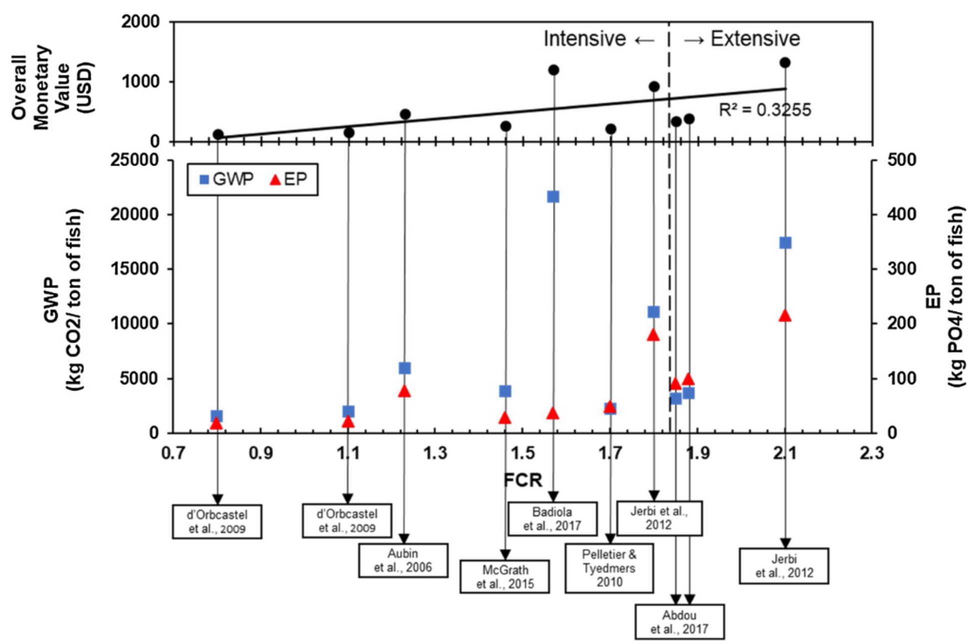

Fig. 4. GWP and EP environmental impacts, and the associated overall monetary value of impacts based on FCRs. data are obtained from the investigated studies with reported GWP, EP, and FCR (10 datapoints).

Laurent, 2019).

As expected, the electricity and fuel use by intensive systems was consistently higher than in extensive systems. Intensive systems have greater pumping and aeration requirements resulting in greater global impacts of cumulative energy demand/energy use and global warming potential (Badiola et al., 2018). In IMTA systems, greater production capacity can potentially moderate these impacts. This potential is illustrated by the reduced energy use at higher production densities with simultaneous production of two fish species in Mungkung et al. (2013). In addition, changing the electricity source can dramatically reduce the environmental impact of intensive RAS (Ayer and Tyedmers, 2009; Samuel-Fitwi et al., 2013; Badiola et al., 2018). Greater development and use of renewable energy sources will decrease the carbon emissions of intensive systems (IEA, 1998; Proksch et al., 2019; Ghamkhar et al., 2020).

Unlike energy, the additional infrastructure attributed to intensive systems does not have a large environmental impact. In the studies that reported infrastructure as a separate process, the environmental impacts were negligible compared to the other impact contributors (i.e. < 13 %). It is common for infrastructure or capital goods to be excluded from a LCA. Buildings are considered to have long lifespans and after their contribution is divided by the building's total lifespan the environmental impact is insignificant (Morais and Delerue-Matos, 2010). Despite the frequent exclusion, Frischknecht et al. (2007) looked at the impacts of capital goods and found that they can have a significant impact on certain impact categories. As such, capital goods should not be excluded without consideration and proper justification for exclusion (Bohnes et al., 2018; Ghamkhar et al., 2020). Several of the reviewed studies indicated that infrastructure did not contribute significantly; however, these studies did not include assumptions about infrastructure lifespan. Exclusion of infrastructure in future aquaculture studies should be considered carefully and will depend on anticipated lifespan of the production system (Ghamkhar et al., 2020).

Similar to infrastructure, the impact category land use also had negligible impacts in intensive systems. The area occupied by tanks and water treatment equipment in intensive systems is much smaller than the area required to produce feed products. Extensive aquaculture requires more on-farm land use due to the increased area needed for pond construction and lower yields. When compared to other protein sources, intensive aquaculture production has fewer land use impacts on a kg live-weight basis. A comparison of pork, poultry, beef, and fish when normalized to m^2/kg edible product indicated fish in RAS to have the lowest land use (Table 3).

Similar to intensive systems, off-farm land use requirements of other protein sources are attributed to feed production (Thomassen et al., 2008). Poultry, beef, and pork rely on similar agricultural feed products as those used to supplement fish meal in aquaculture feeds (Ellingsen and Aanondsen, 2006). Changing the aquaculture feed composition to include more plant derived ingredients could increase the land use requirements of aquaculture production. It could also increase competition for land use with other protein sources due to reliance on the same ingredients (despite potential impact decrease in other categories such as NPPU). In contrast, extensive aquaculture systems require less supplemental feed and indirectly compete less for plant derived feed ingredients; however, extensive systems could compete directly with other protein sources due to the large on-farm area requirements.

Water use is a unique impact factor considered in several of the reviewed papers. Intensive RAS systems utilize water more efficiently and therefore had lower water use impacts than flow-through or extensive aquaculture systems. Of the papers reviewed, two studies accounted for agricultural irrigation (indirect use) and found irrigation contributed significantly to water use (Mungkung et al., 2013; Henriksson et al., 2017). The exclusion of irrigation for feed ingredients by studies on intensive aquaculture systems potentially ignores a large water requirement. Commercial feeds used in intensive systems with a high quantity of plant derived ingredients will have lower NPPU impacts at the risk of greater water use impacts. The agricultural industry is one of the largest users of fresh water resources and most of the grains produced go into animal feeds (Goodland, 1997). If aquaculture feeds incorporate more agriculturally produced plant ingredients, it could potentially increase the water use of those systems placing more stress on limited water supplies. To properly compare water use of an intensive RAS and extensive pond system the water use in feed production must be considered within the system's boundary. Incorporation of the irrigation water for feed production could minimize difference in water use between intensive and extensive systems. For this reason, as with feed and energy, it could be beneficial to integrate aquaculture systems with additional products (Bibbiani et al., 2018; Wu et al., 2019). Increased production per m^3 of water could mitigate indirect agriculture-related water use.

Table 3
Comparison of land use (m^2) results from LCA studies.

Study	System	Functional Unit (FU)	m^2/FU	m^2/kg edible product*
Pork**				
Williams et al. (2006)	Heavier finishing	1 ton dead weight	6900	9.8
Williams et al. (2006)	Indoor breeding	1 ton dead weight	7300	10.3
Williams et al. (2006)	Outdoor breeding	1 ton dead weight	7500	10.6
Williams et al. (2006)	Conventional	1 ton dead weight	7400	10.5
Poultry**				
Williams et al. (2006)	Conventional	1 ton dead weight	6400	8.0
Williams et al. (2006)	Free range	1 ton dead weight	7300	11.9
Beef**				
Williams et al. (2006)	100 % sucker	1 ton dead weight	38,500	49.2
Williams et al. (2006)	Lowland	1 ton dead weight	22,800	29.2
Williams et al. (2006)	Hill and upland	1 ton dead weight	24,100	30.8
Williams et al. (2006)	Non-organic	1 ton dead weight	23,000	29.4
Fish**				
Jerbi et al. (2012)	Cascade flow-through	1 ton live fish weight	4940	9.9
Jerbi et al. (2012)	Traditional flow-through	1 ton live fish weight	4260	8.5
d'Orbcastel et al. (2009)	RAS, FCR 0.8	1 ton fish	2097	4.2
d'Orbcastel et al. (2009)	RAS, FCR 1.1	1 ton fish	2752	5.5
Wilfart et al. (2013)	RAS	1 ton fish	740	1.5
Wilfart et al. (2013)	Semi-extensive pond	1 ton fish	30,897	61.8
Wilfart et al. (2013)	Extensive pond	1 ton fish	56,750	114
Abdou et al. (2017)	Sea-cage (intensive / seabass)	1 ton fish	1336	2.6
Abdou et al. (2017)	Sea-cage (intensive / seabream)	1 ton fish	1369	2.7
Henriksson et al. (2017)	Conventional (intensive / tilapia)	1 ton fish	1199	2.2

* kg edible product for pork, poultry, and beef calculated based on information in De Vries and de Boer (2010); kg edible product for fish based on assumption of 0.5 kg edible product/ kg live weight (Iversen, 1996).

** Data on pork, poultry, and beef from (De Vries and de Boer, 2010). Data on fish based on studies in this review.

While the assessment of water use in the reviewed papers is useful as a baseline comparison between systems, they are extremely simplified. The studies only consider direct quantity of water flowing into the system. As such, the assessments lack distinction between types of water used (blue, green, or grey), consumptive and non-consumptive uses, and spatially relevant scarcity (Ridoutt and Pfister, 2010, 2013). A new method to describe both consumptive and degradative water use, while incorporating an indicator of global water stress is developed for LCA (Ridoutt and Pfister, 2013). Future research on aquaculture should include this new method or even the commonly used Water Footprint Network method as described by (Hoekstra et al., 2011), which includes indirect water use to provide more robust measures of water use.

Despite possible limitations in the water use category, increased water efficiency resulted in lower eutrophication potentials. Extensive systems that rely on pond fertilization have greater direct emissions due to on-farm production. In addition to greater direct emissions, the lower yields in an extensive system resulted in a greater eutrophication potential per unit mass of ultimate product, compared to the highly productive intensive systems (Thomassen et al., 2008; Bibbiani et al., 2018; Bohnes et al., 2018). Furthermore, some extensive systems also supplement with commercial feeds thereby increasing indirect emissions from plant derived feed ingredients (Pelletier and Tyedmers, 2010; Chary et al., 2020). In contrast, intensive systems are the result of a historical focus on reducing local water quality and ecological impacts. The low eutrophication potential of RAS is evidence to support the success of this movement. Instead of direct emissions, eutrophication potential is largely due to the off-farm impacts of energy production and feed production. Therefore, further reductions in eutrophication potential will come from reducing the impacts of feed and energy with improved FCRs, alternative feed ingredients, and alternative energy sources, or the elimination of all waste discharge. Such zero-emission RAS are currently being developed that include IMTA or additional treatment systems (Van Rijn, 2013; Chary et al., 2020).

While zero-emission RAS, specifically IMTA, have great potential to reduce the environmental impact of aquaculture systems (Czyrnek-Delêtre et al., 2017; Ianchenko and Proksch, 2019; Proksch et al., 2019; Ghamkhar et al., 2020), future research is needed to quantitatively evaluate these new systems. It remains in question how the incorporation of additional products will change the environmental impact when evaluated through LCA. In addition, methods to address allocation in multi-output IMTA systems has yet to be studied. In this review, seven papers included allocation and of those only two applied the system expansion method. Considering the inevitable allocation issues in IMTA and its limited use in aquaculture studies, the use of system expansion to address allocation in both IMTA and aquaculture are potential research areas.

Future LCAs on zero-emission aquaculture systems, freshwater and marine, will be needed to clarify the advantages and disadvantages of multiple products and its associated water treatment in terms of environmental impact. Just as there was a possible burden shift moving from extensive to intensive aquaculture systems a more in-depth assessment of zero-emission systems may uncover trade-offs to integration.

5. Conclusion

A comparison of different production systems, with a focus on the differences between intensive land-based systems and extensive pond systems, showed an improvement of overall environmental performance, with a possibility of burden shifting when moving to more intensive aquaculture systems. Intensive systems are often considered to have fewer negative environmental impacts than extensive systems, specifically less water pollution and total water use. Exploration of these environmental impacts through the LCA lens provided support for these claims about intensive aquaculture. It also showed that other impacts, such as cumulative energy demand/energy use and NPPU, are greater. In areas where electricity is predominately supplied by fossil fuels, the greater energy requirements correspond with greater carbon emissions. Facilities located in areas, such as Europe, that have access to renewable energy sources benefit from a reduction in carbon emissions despite greater energy requirements. The future of intensive land-based aquaculture development in the United States, which does not have a strong renewable energy market, nor has it established a federal renewable energy policy to encourage such a market, is at a distinct disadvantage due to the lack of renewable energy sources.

In addition to greater access to renewable energy sources, development of sustainable fish feed and the improvement of feed conversion efficiencies will reduce the environmental impacts of aquaculture. Aquaculture feed is well known to have large biotic resource and energy requirements. While the movement from extensive to intensive aquaculture resulted in an improvement of FCRs, fish can only incorporate a certain percentage of the nutrients in feed. Alternative aquaculture systems, which improve the total nutrient uptake and increase total yields thereby reducing impacts through greater production per unit of feed, water, and energy, are needed to further reduce the impacts of aquaculture.

This review demonstrates that while intensive aquaculture systems have greatly reduced negative, local environmental impacts; many negative, global environmental impacts may still remain without applying case-specific mitigation strategies. The achievement of sustainable aquaculture production will likely come from both improved technologies and a careful balance between local and global environmental impacts through management of production intensities.

Author statement

All authors have contributed to this work.

Declaration of Competing Interest

The authors report no declarations of interest.

Acknowledgements

This research has received support by funding from National Science Foundation#1942110, Florida Sea Grant#SI-2014-0006; the U.S. Department of Education Graduate Assistants in Area of National Need (GAANN) Fellowship, project #P200A090162. Support was also provided by the National Science Foundation S-STEM Grant#0965743. Any opinions, findings, and conclusions or recommendations in this material are those of the author(s) and do not necessarily reflect the views of the sponsors. The authors appreciate all the support and insights provided by colleagues and friends upon this collaborative work.

References

Abdou, K., Aubin, J., Romdhane, M.S., Le Loc'h, F., Lasram, F.B.R., 2017. Environmental assessment of seabass (Dicentrarchus labrax) and seabream (Sparus aurata) farming from a life cycle perspective: a case study of a Tunisian aquaculture farm. Aquaculture 471, 204–212.
Andersson, K., 2000. LCA of food products and production systems. Int. J. Life Cycle Ass. 5, 239.
Aubin, J., Papatryphon, E., Van der Werf, H., Petit, J., Morvan, Y., 2006. Characterisation of the environmental impact of a turbot (Scophthalmus maximus) re-circulating production system using Life Cycle Assessment. Aquaculture 261, 1259–1268.
Aubin, J., Papatryphon, E., Van der Werf, H., Chatzifotis, S., 2009. Assessment of the environmental impact of carnivorous finfish production systems using life cycle assessment. J. Cleaner Prod. 17, 354–361.
Avadí, A., Fréon, P., 2013. Life cycle assessment of fisheries: a review for fisheries scientists and managers. Fish. Res. 143, 21–38.
Ayer, N.W., 2007. The biophysical costs of technology: assessing the environmental impacts of alternative salmonid culture systems in Canada using Life Cycle Assessment. ProQuest.
Ayer, N.W., Tyedmers, P.H., 2009. Assessing alternative aquaculture technologies: life cycle assessment of salmonid culture systems in Canada. J. Cleaner Prod. 17, 362–373.

Badiola, M., Mendiola, D., Bostock, J., 2012. Recirculating aquaculture systems (RAS) analysis: Main issues on management and future challenges. Aquacult. Eng. 51, 26–35.
Badiola, M., Basurko, O., Gabiña, G., Mendiola, D., 2017. Integration of energy audits in the Life Cycle Assessment methodology to improve the environmental performance Assessment of Recirculating Aquaculture Systems. J. Cleaner Prod. 157, 155–166.
Badiola, M., Basurko, O., Piedrahita, R., Hundley, P., Mendiola, D., 2018. Energy use in recirculating aquaculture systems (RAS): a review. Aquacult. Eng. 81, 57–70.
Barrington, K., Chopin, T., Robinson, S., 2009. Integrated multi-trophic aquaculture (IMTA) in marine temperate waters. Integrated mariculture: a global review. In: FAO Fisheries and Aquaculture Technical Paper, 529, pp. 7–46.
Basto-Silva, C., Guerreiro, I., Oliva-Teles, A., Neto, B., 2019. Life cycle assessment of diets for gilthead seabream (Sparus aurata) with different protein/carbohydrate ratios and fishmeal or plant feedstuffs as main protein sources. Int. J. Life Cycle Ass. 1–12.
Bibbiani, C., Fronte, B., Incrocci, L., Campiotti, C.A., 2018. Life cycle impact of industrial aquaculture systems: a review. Calitatea 19, 67–71.
Biermann, G., Geist, J., 2019. Life cycle assessment of common carp (Cyprinus carpio L.)– A comparison of the environmental impacts of conventional and organic carp aquaculture in Germany. Aquaculture 501, 404–415.
Bohnes, F.A., Laurent, A., 2019. LCA of aquaculture systems: methodological issues and potential improvements. Int. J. Life Cycle Ass. 24, 324–337.
Bohnes, F.A., Hauschild, M.Z., Schlundt, J., Laurent, A., 2018. Life cycle assessments of aquaculture systems: a critical review of reported findings with recommendations for policy and system development. Rev. Aquacult.
Boxman, Suzanne E., Zhang, Qiong, Bailey, Donald, Trotz, Maya A., 2017. Life cycle assessment of a commerical-scale freshwater aquaponic system. Environ. Eng. Sci. 34 (5), 299–311.
Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2002. Life cycle impact assessment of land use based on the hemeroby concept. Int. J. Life Cycle Ass. 7, 339.
Buric, M., Bláhovec, J., Kouril, J., 2014. A simple and effective recirculating hatchery for salmonids. J. Aquac. Res. Dev. 5, 2.
Campbell, P.K., Beer, T., Batten, D., 2011. Life cycle assessment of biodiesel production from microalgae in ponds. Bioresour. Technol. 102, 50–56.
Carter, C., Bransden, M., Lewis, T., Nichols, P., 2003. Potential of thraustochytrids to partially replace fish oil in Atlantic salmon feeds. Mar. Biotechnol. 5, 480–492.
Cederberg, C., Stadig, M., 2003. System expansion and allocation in life cycle assessment of milk and beef production. Int. J. Life Cycle Ass. 8, 350–356.
Chary, K., Aubin, J., Sadoul, B., Fiandrino, A., Covès, D., Callier, M.D., 2020. Integrated multi-trophic aquaculture of red drum (Sciaenops ocellatus) and sea cucumber (Holothuria scabra): assessing bioremediation and life-cycle impacts. Aquaculture 516, 734621.
Cooper, J.S., 2003. Life-cycle assessment and sustainable development indicators. J. Ind. Ecol. 7, 12–15.
Council, D.P., 2013. Technical Support Document:-Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis-Under Executive Order 12866. Environmental Protection Agency.
Curran, M.A., 2006. Scientific Applications International Corporation. Life-Cycle Assessment: Principles and Practice. National Risk Management Research Laboratory, Office of Research and ….
Czyrnek-Delêtre, M.M., Rocca, S., Agostini, A., Giuntoli, J., Murphy, J.D., 2017. Life cycle assessment of seaweed biomethane, generated from seaweed sourced from integrated multi-trophic aquaculture in temperate oceanic climates. Appl. Energy 196, 34–50.
d'Orbcastel, E.R., Blancheton, J.-P., Aubin, J., 2009. Towards environmentally sustainable aquaculture: comparison between two trout farming systems using Life Cycle Assessment. Aquacult. Eng. 40, 113–119.
Davidson, J., Barrows, F.T., Kenney, P.B., Good, C., Schroyer, K., Summerfelt, S.T., 2016. Effects of feeding a fishmeal-free versus a fishmeal-based diet on post-smolt Atlantic Salmon Salmo salar performance, water quality, and waste production in recirculation aquaculture systems. Aquacult. Eng. 74, 38–51.
De Vries, M., de Boer, I.J., 2010. Comparing environmental impacts for livestock products: a review of life cycle assessments. Livestock Sci. 128, 1–11.
Efole Ewoukem, T., Aubin, J., Tomedi Eyango Tabi, M., Mikolasek, O., Corson, M.S., Tchoumboue, J., Van Der Werf, H.M., Ombredane, D., 2010. Environmental Impacts of Farms Integrating Aquaculture and Agriculture in Cameroon.
Ellingsen, H., Aanondsen, S.A., 2006. Environmental impacts of wild caught cod and farmed salmon-a comparison with chicken (7 pp). Int. J. Life Cycle Ass. 11, 60–65.
FAO, 2018. The State of world fisheries and aquaculture, 2018. Food Agric. Org.
Frischknecht, R., Jungbluth, N., Althaus, H.-J., Hischier, R., Doka, G., Bauer, C., Dones, R., Nemecek, T., Hellweg, S., Humbert, S., 2007. Implementation of Life Cycle Impact Assessment Methods. Data v2. 0 (2007). Ecoinvent Report No. 3. Ecoinvent Centre.
Ghamkhar, R., Hicks, A., 2020. Comparative environmental impact assessment of aquafeed production: sustainability implications of forage fish meal and oil free diets. Resour. Conserv. Recycl. 161, 104849.
Ghamkhar, R., Hartleb, C., Wu, F., Hicks, A., 2020. Life cycle assessment of a cold weather aquaponic food production system. J. Cleaner Prod., 118767
Goodland, R., 1997. Environmental sustainability in agriculture: diet matters. Ecol. Econ. 23, 189–200.
Granada, C., 2015. De mónadas y sustantividades o Leibniz y Zubiri. Pensamiento 266, pp. 251–276.
Gronroos, J., Seppala, J., Silvenius, F., Makinen, T., 2006. Life cycle assessment of Finnish cultivated rainbow trout. Boreal Environ. Res. 11, 401.
Guinée, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. Int. J. Life Cycle Ass. 7, 311.
Hasan, M., Soto, D., 2017. Improving Feed Conversion Ratio and Its Impact on Reducing Greenhouse Gas Emissions in Aquaculture. Improving Feed Conversion Ratio and Its Impact on Reducing Greenhouse Gas Emissions in Aquaculture.
Henriksson, P.J., Guinée, J.B., Kleijn, R., de Snoo, G.R., 2012. Life cycle assessment of aquaculture systems—a review of methodologies. Int. J. Life Cycle Ass. 17, 304–313.
Henriksson, P.J., Dickson, M., Allah, A.N., Al-Kenawy, D., Phillips, M., 2017. Benchmarking the environmental performance of best management practice and genetic improvements in Egyptian aquaculture using life cycle assessment. Aquaculture 468, 53–59.
Hoekstra, A.Y., Chapagain, A.K., Mekonnen, M.M., Aldaya, M.M., 2011. The Water Footprint Assessment Manual: Setting the Global Standard. Routledge.
Ianchenko, A., Proksch, G., 2019. Urban food systems: applying life cycle assessment in built environments and aquaponics. Build. Technol. Ed. Soc. 2019, 29.
IEA, 1998. Benign Energy? The Environmental Implications of Renewables. OECD Publishing.
ISO, 2008. ISO 15270: Plastics – Guidelines for the Recovery and Recycling of Plastics Waste.
ISO-Norm, I., 2006. Environmental Management—Life Cycle Assessment—Principles and Framework ISO 14040: 2006. ISO, Geneva, Switzerland.
Iversen, E.S., 1996. Food and Nonfood Fisheries. Living Marine Resources. Springer, pp. 176–204.
Jerbi, M., Aubin, J., Garnaoui, K., Achour, L., Kacem, A., 2012. Life cycle assessment (LCA) of two rearing techniques of sea bass (Dicentrarchus labrax). Aquacult. Eng. 46, 1–9.
Lucas, J.S., Southgate, P.C., Tucker, C.S., 2019. Aquaculture: Farming Aquatic Animals and Plants. John Wiley & Sons.
Mattila, T., Helin, T., Antikainen, R., Soimakallio, S., Pingoud, K., Wessman, H., 2011. Land use in life cycle assessment. Finnish Environ. 24.
McGrath, K.P., Pelletier, N.L., Tyedmers, P.H., 2015. Life cycle assessment of a novel closed-containment salmon aquaculture technology. Environ. Sci. Technol. 49, 5628–5636.
Morais, S.A., Delerue-Matos, C., 2010. A perspective on LCA application in site remediation services: critical review of challenges. J. Hazard. Mater. 175, 12–22.
Mungkung, R., Aubin, J., Prihadi, T.H., Slembrouck, J., van der Werf, H.M., Legendre, M., 2013. Life cycle assessment for environmentally sustainable aquaculture management: a case study of combined aquaculture systems for carp and tilapia. J. Cleaner Prod. 57, 249–256.
Naylor, R.L., Goldburg, R.J., Primavera, J.H., Kautsky, N., Beveridge, M.C., Clay, J., Folke, C., Lubchenco, J., Mooney, H., Troell, M., 2000. Effect of aquaculture on world fish supplies. Nature 405, 1017.
Neori, A., Chopin, T., Troell, M., Buschmann, A.H., Kraemer, G.P., Halling, C., Shpigel, M., Yarish, C., 2004. Integrated aquaculture: rationale, evolution and state of the art emphasizing seaweed biofiltration in modern mariculture. Aquaculture 231, 361–391.
Papatryphon, E., Petit, J., Kaushik, S.J., van der Werf, H.M., 2004. Environmental impact assessment of salmonid feeds using life cycle assessment (LCA). AMBIO 33, 316–323.
Pauly, D., Christensen, V., 1995. Primary production required to sustain global fisheries. Nature 374, 255–257.
Pelletier, N., Tyedmers, P., 2007. Feeding farmed salmon: Is organic better? Aquaculture 272, 399–416.
Pelletier, N., Tyedmers, P., 2010. Life cycle assessment of frozen tilapia fillets from Indonesian lake-based and pond-based intensive aquaculture systems. J. Ind. Ecol. 14, 467–481.
Pelletier, N., Tyedmers, P., Sonesson, U., Scholz, A., Ziegler, F., Flysjo, A., Kruse, S., Cancino, B., Silverman, H., 2009. Not All Salmon Are Created Equal: Life Cycle Assessment (LCA) of Global Salmon Farming Systems. ACS Publications.
Philis, G., Ziegler, F., Gansel, L.C., Jansen, M.D., Gracey, E.O., Stene, A., 2019. Comparing life cycle assessment (LCA) of Salmonid aquaculture production systems: Status and perspectives. Sustainability 11, 2517.
Phong, L., De Boer, I., Udo, H., 2011. Life cycle assessment of food production in integrated agriculture–aquaculture systems of the Mekong Delta. Livestock Sci. 139, 80–90.
Pizzol, M., Weidema, B., Brandão, M., Osset, P., 2015. Monetary valuation in life cycle assessment: a review. J. Cleaner Prod. 86, 170–179.
Proksch, G., Ianchenko, A., Kotzen, B., 2019. Aquaponics in the Built Environment. Aquaponics Food Production Systems. Springer, pp. 523–558.
Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environ. Change 20, 113–120.
Ridoutt, B.G., Pfister, S., 2013. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. Int. J. Life Cycle Ass. 18, 204–207.
Samuel-Fitwi, B., Nagel, F., Meyer, S., Schroeder, J., Schulz, C., 2013. Comparative life cycle assessment (LCA) of raising rainbow trout (Oncorhynchus mykiss) in different production systems. Aquacult. Eng. 54, 85–92.
Stickney, R.R., 1994. Principles of Aquaculture. John Wiley and Sons, Inc.
Stokes, J., Horvath, A., 2006. Life cycle energy assessment of alternative water supply systems (9 pp). Int. J. Life Cycle Ass. 11, 335–343.
Temizel-Sekeryan, S., Wu, F., Hicks, A.L., 2020. Life Cycle Assessment of Struvite Precipitation from Anaerobically Digested Dairy Manure: A Wisconsin Perspective. Integrated Environmental Assessment and Management.
Thomassen, M.A., van Calker, K.J., Smits, M.C., Iepema, G.L., de Boer, I.J., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agric. Syst. 96, 95–107.
Van Rijn, J., 2013. Waste treatment in recirculating aquaculture systems. Aquacult. Eng. 53, 49–56.

Weidema, B.P., 2009. Using the budget constraint to monetarise impact assessment results. Ecol. Econ. 68, 1591–1598.

Wilfart, A., Prudhomme, J., Blancheton, J.-P., Aubin, J., 2013. LCA and emergy accounting of aquaculture systems: towards ecological intensification. J. Environ. Manage. 121, 96–109.

Williams, A., Audsley, E., Sandars, D., 2006. Determining the Environmental Burdens and Resource Use in the Production of Agricultural and Horticultural Commodities: Defra Project Report IS0205. Zu finden in: http://randd. defra. gov. uk/Default. aspx.

Wirza, R., Nazir, S., 2020. Urban aquaponics farming and cities-a systematic literature review. Rev. Environ. Health 1.

Wu, F., Ghamkhar, R., Ashton, W., Hicks, A.L., 2019. Sustainable Seafood and Vegetable Production–Aquaponics as a Potential Opportunity in Urban Areas. Integrated Environmental Assessment and Management.

Wu, F., Zhou, Z., Temizel-Sekeryan, S., Ghamkhar, R., Hicks, A.L., 2020. Assessing the environmental impact and payback of carbon nanotube supported CO2 capture technologies using LCA methodology. J. Cleaner Prod., 122465

Yacout, D.M., Soliman, N.F., Yacout, M., 2016. Comparative life cycle assessment (LCA) of tilapia in two production systems: semi-intensive and intensive. Int. J. Life Cycle Ass.Assessment 21, 806–819.