Doctoral theses at NTNU, 2021:335

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Doctoral thesis

NTNU
Norwegian University of Science and Technology
Thesis for the Degree of
Philosophiae Doctor
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Department of Biological Sciences Ålesund

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Summary

The development of food production systems and the adoption of diets with lower environmental burdens are critical to mitigate the threats from climate change and the erosion of biodiversity and ecosystems. Many consider seafood to be a viable alternative source of animal protein to the most polluting types of ruminant production, such as cattle and sheep. Farmed salmon is a popular finfish providing an alternative to meat, appreciated for its taste, the quality of its proteins, and its sources of marine omega 3. Despite significantly lower life cycle impacts than most land-based animal production, the salmon aquaculture industry faces substantial environmental challenges. In Norway, large production volumes concentrated in open marine cages led to the chronic contaminations of coastal areas by viruses and parasites. This reduces production efficiency, fish welfare and threatens the stocks of wild salmon. Permanent sea lice infestations in net pens force farmers to use new delousing methods, exacerbating the situation. The Norwegian aquaculture industry is unable to increase its production output sustainably and finds itself at a crossroads. Farmers are investing in alternative land-based and sea-based aquaculture systems without a comprehensive understanding of the environmental tradeoffs involved.

This work intends to improve our understanding of the environmental strengths and weaknesses of salmon aquaculture systems. I used Life Cycle Assessment (LCA) in most of my research to account for environmental impacts generated through life cycles and value chains. First, I reviewed the salmon LCA literature and applied a simple parametric statistical protocol to compare the LCA results of different salmon systems across studies. Then, I conducted LCA of the biological, mechanical, and chemical lice treatments used by the Norwegian aquaculture industry. The rationale for this work was the recent transformation of the treatment mix and the exclusion of treatments' impacts from the LCA of net pen salmons. Finally, I used the LCA of warmwater fish RAS farming in Sweden from Bergman and colleagues and an innovative winter fallowing to control sea-lice infestations in net pens suggested by Stene and colleagues to discuss the tradeoffs and future of aquaculture systems in Norway.

Despite small data samples and multiple confounding factors, the cross-study statistical comparison was successful for some portions of the data. I demonstrate that (1) sea-based systems require significantly less energy than land-based systems, (2) land-based systems have a significantly lower feed conversion ratio than sea-based systems, and (3) closed systems likely have a significantly lower eutrophying potential than open systems. Norwegian farmers' current lice treatment mix adds significant life cycle impacts to net pen salmons, especially for the carbon, marine toxicity, and energy footprints. The main impact drivers are the increased salmon mortality, the fuel use from ships, the production of hydrogen peroxide, and the construction of mechanical treatment units. However, preliminary observations suggest that adding the treatment impacts to the life cycle impacts of net pen salmons will have a negligible effect on system comparisons. Regarding the LCA methodology itself, I argue in favor of more data reusability and interoperability using the lice treatments LCA to showcase the possibility of sharing openly human and machine-readable inventories while respecting confidentiality agreements. I also highlight the limitations of LCA for the comparison of aquaculture systems, particularly with regards to impacts on biodiversity, ecosystems, and fish welfare. Finally, based on the current state of knowledge, I argue against the large-scale development of land-based, offshore, and closed sea-based systems envisioned by some stakeholders in Norway. I recommend testing nature’s strategy suggested by Stene and colleagues to mitigate sea lice challenges and improve the environmental profile of open sea-based systems. A low technology solution like this could allow the industry to increase its production output by keeping more fish in the cages alive.

Preface & acknowledgment

This thesis is submitted in partial fulfillment of the requirements for the degree of Philosophiae Doctor (Ph.D.) at the Norwegian University of Science and Technology (NTNU). This degree has been undertaken at the Department of Biological Sciences Ålesund, under the faculty of Natural Sciences, between September 2017 and June 2021. This Ph.D. project was conducted in collaboration with RISE – The Swedish Research Institute and the Norwegian Veterinary Institute and funded by the NTNU Sustainability Strategic Research Area.

First and foremost, I would like to extend my profound gratitude to my supervisors Anne Stene, Lars Christian Gansel, Friederike Ziegler, and Mona Dverdal Jansen. Anne, thank you for your steady guidance and for giving me much freedom throughout this Ph.D. You supported me with deep knowledge of the salmon aquaculture industry and helped link fish biology and LCA. Thanks, Lars, for your availability, your good advice on scientific writing, and our discussions about sustainability. You really do your best to integrate your Ph.D. students both professionally and socially. A special thanks to Friederike and Mona for accepting to jump onboard this Ph.D. project. Friederike, I think I can say without a doubt that this LCA-based thesis could not have been done without your participation. I keep great memories of my two visits to RISE. Thank you very much for introducing me to your research environment and giving so much of your time supervising my work. Many thanks to you, Mona, for consistently supporting this work. You provided well-needed guidance to navigate statistic testing and always gave me pragmatic and efficient feedbacks along the way.

I would also like to acknowledge Annik Magerholm Fet. Thank you for guiding me from my master thesis into a Ph.D. and giving me opportunities to gain experience through research projects, standardization work, and international reports. A special thanks go to Tove Havnegjerde and all my colleagues of the department of biology in Ålesund. Working next to you during the last four years has been a fulfilling professional and personal experience.

Finally, I want to express all my love and gratitude to Nina Benedicte Aubert. You helped me in so many ways throughout this Ph.D. journey. Thank you for your intelligent advice, for listening, caring, and giving me room to work and write.

Abbreviations

CC	Climate Change
CED	Cumulative Energy Demand
FCR	Feed Conversion Ratio
FTS	Flow-Through System
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
LU	Land Use
MET	Marine EcoToxicity
MEU	Marine EUtrophication
RAS	Recirculating Aquaculture System
WU	Water Use

List of publications

- Primary papers: main contribution of the candidate.

Paper I

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*(9), 2517. https://doi.org/10.3390/su11092517. Contribution = conceived/designed the study, performed the review, and drafted the article.

Paper II

Philis, G., Ziegler, F., Jansen, M. D., Gansel, L. C., Hornborg, S., Aas, G. H., & Stene, A. Quantifying environmental impacts of cleaner fish used as sea lice treatments in salmon aquaculture with life cycle assessment. *J Ind Ecol*. 2021; 114. https://doi.org/10.1111/jiec.13118. Contribution = conceived/designed the study, performed the life cycle assessment, and drafted the article.

Paper III

Philis, G., Ziegler, F., Snåre M. W., Jansen, M. D., Gansel, L. C., & Stene, A. Quantifying environmental impacts of sea lice treatments in salmon aquaculture with life cycle assessment (submitted to Journal of Industrial Ecology). Contribution = conceived/designed the study, performed the life cycle assessment, and drafted the article.

❖ Secondary papers: providing perspectives to the discussion and conclusion.

Paper IV

Bergman, K., Henriksson, P. J. G., Hornborg, S., Troell, M., Borthwick, L., Jonell, M., Philis, G., & Ziegler, F. (2020). Recirculating Aquaculture Is Possible without Major Energy Tradeoff: Life Cycle Assessment of Warmwater Fish Farming in Sweden. *Environmental science & technology*. https://doi.org/10.1021/acs.est.0c01100. Contribution = provided perspectives on biological impacts of cage farming and reviewed the manuscript.

Paper V

Stene, A., Fjørtoft, H. B., Hellevik, C., Philis, G. Using nature's strategy to control sea lice infestation in marine salmon cage culture? (Re-submitted to Aquaculture Environment Interactions). Contribution = provided data and calculation used in figure 1 and reviewed the manuscript.

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1 Introduction

1.1 Sustainable food production

In March 2021, a paper published in Nature Food concluded that a third of all anthropogenic greenhouse gas emissions come from the world food systems (Crippa et al., 2021). It is also well documented that agriculture, the foremost component of this system, is by far the main driver of ecosystem degradation and biodiversity loss (Bartlett et al., 2016; Benton et al., 2021). Such findings blatantly illustrate how vital a transformation of this sector is to mitigate climate change and the loss of the earth's ecosystems and biodiversity. Structural societal trends like population growth and the rising incomes in some middle-class populations drive impacts in the wrong direction by increasing the overall demand for foods, particularly for those rich in animal proteins (Godfray & Garnett, 2014; Wu et al., 2014).

Shifting to more sustainable foods is a complex and challenging transformation requiring drastic changes from production and consumption (Poore & Nemecek, 2018). Environmental and health research is now converging, demonstrating the overarching benefits of reducing the proportion of animal products and increasing the share of plants, particularly in the western diet (Rust et al., 2020; Willett et al., 2019). While a shift to more plant-based diets is inescapable to make a significant difference, several researchers have shown that seafood could be an excellent alternative to meat to reduce the overwhelming environmental impacts of this food category, particularly from ruminant livestock (Hallström et al., 2019; Hilborn et al., 2018; Scarborough et al., 2014).

Replacing land-based animal products with seafood comes with advantages and challenges (Costello et al., 2020). Capture output is stagnating, and many scenarios predict a decreasing production from fisheries because of overfishing and marine pollution issues (FAO, 2018). If these trends continue, we expect that mariculture will drive the growth of the seafood sector (Kobayashi et al., 2015; Olsen, 2015). In Norway, there is an apparent gain of interest to develop industrial processes to farm low trophic species such as seaweed and mussels (Handå, 2012; Stévant et al., 2017). However, the mariculture strategy remains centered on finfish aquaculture, particularly of salmon, due to the substantial economic and social importance of this industry for the country (Johansen et al., 2019). Salmon is a carnivorous finfish particularly appreciated for its palatability and taste, source of marine omega-3, and mild-orange colored meat containing high-quality proteins (Farmer et al., 2000; Sprague et al., 2016). Although linked to environmental challenges, its production is also associated with low freshwater footprints and economic Feed Conversion Ratio (FCR) compared to meat agriculture products (Torrissen et al., 2011).

1.2 Norwegian salmon: trends and challenges

Norway is one of the few western countries with a large aquaculture industry, producing alone 48% of all salmonids farmed in marine waters (FAO, 2021). Production along its shores is ideal for salmonid production: an extensive coast with clean and cool waters, sheltered by fjords, and temperated by Gulf Stream currents. The Norwegian salmonid production is dominated at 94% by Atlantic salmon (*Salmo salar*), with the 6% remaining coming from Rainbow trout (*Oncorhynchus mykiss*) farming (NDF, 2020a). The volume of salmon[1] produced increased rapidly between 1998 and 2012 before oscillating close to 1.4 million tons of fish in more recent years (Figure 1). The total value of salmon slaughtered kept increasing substantially over the 2012-2019 period, despite the production output remaining stable. This value surge suggests a sustained price increase of the commodity.

The current production stagnation is linked to the challenging biological conditions that followed the intensification of farming methods and the significant production increase of the early 2000s (Abolofia et al., 2017; Taranger et al., 2015). Salmon farmers struggle to control some viruses like pancreas disease and infectious salmon anemia as well as parasites like sea lice (*Lepeophtheirus salmonis* and *Caligus elongatus)* (Grefsrud et al., 2019; Hjeltnes et al., 2019). While the pressure from the industry to increase production levels remains strong, Norway made ambitious commitments through the publication of White Paper 16 in 2015, which sets the ground rules for the future sustainable growth of the Norwegian aquaculture industry (NMTIF, 2015).

[1] From this point on, I use the term "salmon" generically to designate the different types of salmonid species.

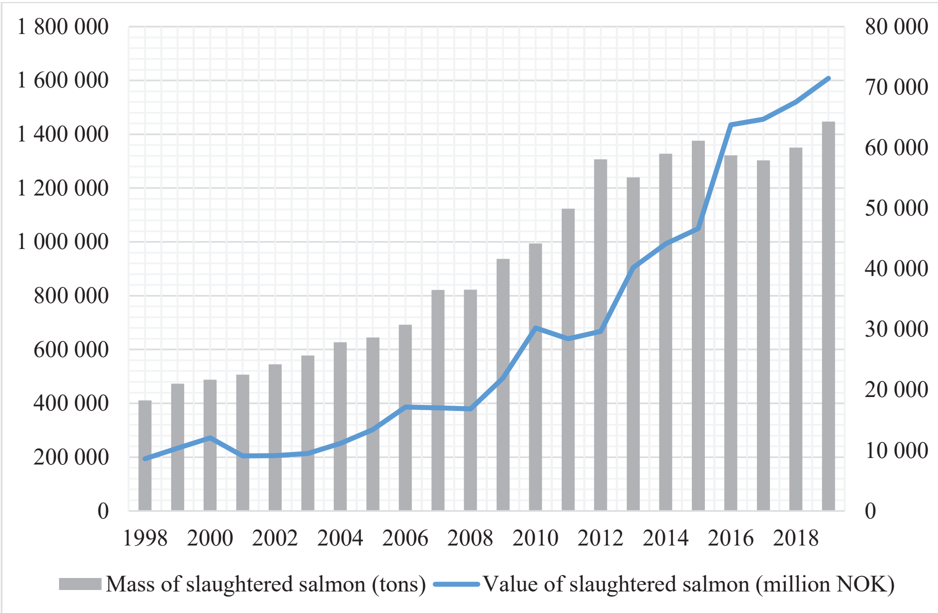

Figure 1. Evolution of the quantity and value of salmonid produced and slaughtered in Norway between 1998 and 2019 (NDF, 2020a). The design of this graph is inspired by Statistics Norway (SSB, 2018).

Following the publication of this roadmap, Norwegian authorities established the traffic light system, dividing Norway into 13 different production zones and regulating changes in biomass production, primarily based on sea lice infestation levels (NMTIF, 2017). The traffic light system require weekly sea lice counts in all net pens and aims to reduce the hazard of sea lice-induced mortality on wild salmonid. When juvenile salmons are wandering out of the Norwegian rivers in the spring, farmers must take mitigating actions to reduce the lice burden if the lice count exceeds 0.2 lice per fish (0.5 during the rest of the year) (NMTIF, 2012). Based on lice infestation levels and the degree of risk for wild salmonids, green, yellow, and red lights are attributed to production zones, defining if biomass production can increase, remain stable or decrease (NMTIF, 2017).

The production costs of Norwegian salmon farmers increasing substantially from around 19 kr/kg fish during 2008-2012 to 28 kr/kg fish between 2013-2019 according to the profitability surveys conducted by the Norwegian Directorate of Fisheries. While costs increased in most categories, there is an apparent surge from expenses relating to fish health, environment, and maintenance (NDF, 2020b). These charges can be directly linked to the increased use of lice treatments that followed the implementation of the sea lice burden limit and significant changes in the treatment mixes since 2012 (BW, 2021b; Liu & Bjelland, 2014).

1.3 Production systems

Sea-based systems: net pens

Circular net pen (1) and rectangular steel farms (2) are the most common production systems used to farm salmonids in marine conditions (Figure 2). Norwegian farmers use the circular net pen in most localities and usually reserve rectangular/square steel cages to sheltered conditions (Fredheim & Langan, 2009). Net pen systems consist of high-density plastic rings, nets, and mooring systems. The circumference of Norwegian net pens reaches up to 157 meters with nets usually 35-meters-deep, hosting a maximum of 200,000 individual fish (Føre et al., 2018).

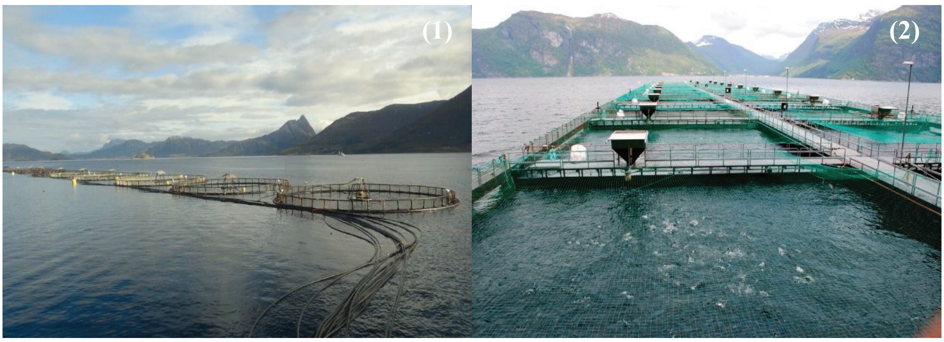

Figure 2. Examples of (1) circular net pen and (2) steel farm used in Norway (Photographs: Øyvind André Haram and Are Kvistad, courtesy of the Norwegian Seafood Federation).

Sea-based systems: alternative concepts

In recent years, innovative sea-based concepts have been studied and developed to address biological challenges affecting net pen production. In Canada, the performances of closed sea-based systems (solid wall and marine floating bag) were compared to conventional net pen (Ayer & Tyedmers, 2009; McGrath et al., 2015). Since 2016, the Norwegian Directorate of Fisheries has delivered several development licenses to closed and partially closed sea-based systems and open offshore solutions. Towards the end of 2019, production licenses for approximately 57,000 tons of biomass have been granted to various salmon farmers (NDF, 2021). These different technologies aim to reduce sea lice infestation levels during grow out production at sea. This could be achieved by either moving the production system offshore, where there are few hosts and consequently less sea lice, or by controlling (partially or totally) the seawater in the system (Figure 3).

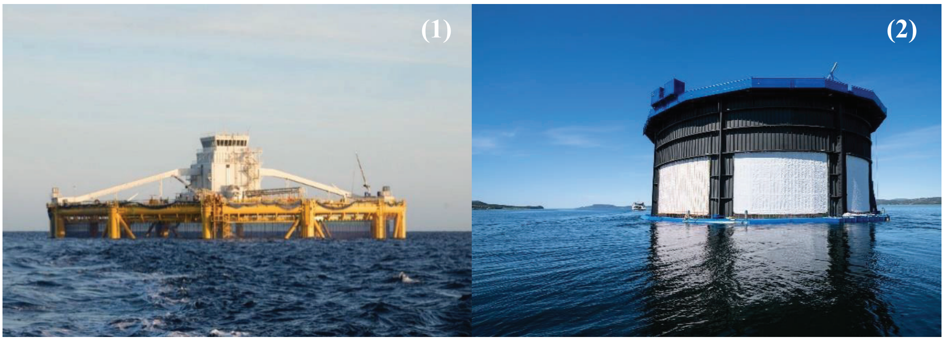

Figure 3. Examples of (1) Salmar's Ocean Farm 1 designed for salmon grow-out in offshore conditions (Photograph: Thor Nielsen, courtesy of the Norwegian Seafood Federation) and (2) Aquatraz semi-closed cage system dragged to a locality by a boat (Photographs: Steinar Johansen, courtesy of Midt-Norsk Havbruk AS).

Land-based systems: Flow-Through System (FTS)

Most Norwegian hatcheries built until the early 2000s were equipped with FTS technology, with a significant proportion of smolt facilities still using this technology today. In FTS systems, freshwater/seawater is derived from a river, a lake, or the sea and is circulated in grow-out tanks before being pumped out, together with the effluents, to the ocean (Bergheim et al., 2009). A typical farm consists of an indoor nursery and water treatment module, and outdoor tanks dedicated to parr-smolt production. Norwegian FTS hatcheries are either single-pass or allowing partial reuse of the water. Water treatment varies according to the type of FTS. Single pass-system only require oxygenation and usually have a minimum water consumption of 0.3 liters per minute per kg fish. Most FTS hatcheries allow partial water recycling, requiring additional treatments like aeration and particle removal and minimum water consumption of 0.15 liter per minute per kg fish (Bergheim et al., 2009).

Land-based systems: Recirculating Aquaculture System (RAS)

Today, most new hatcheries and smolt production facilities built in Norway and several land-based post-smolt projects are betting on RAS technology (EY, 2019). RAS systems offer strictly controlled environments where rearing tanks are placed indoors together with complex treatment modules, pumps, and a heat exchanger (Figure 4). The solid fraction in effluent water (feed spills and feces) is removed by mechanical filtering (e.g., using a 40-micron filter). Water is disinfected with UV and CO2 removed using a degasser. Finally, dissolved effluents like ammonia are converted to nitrite and nitrate using biological filters (Kolarevic et al., 2014). Comprehensive water treatment allows for recycling between 95 to 99.9% of the fresh or seawater used in the system, reducing water requirements to 0.02-0.04 liters per minute per kg fish with this technology (Bergheim et al., 2009).

Figure 4. Juvenile salmon production in a RAS system (Photograph: Smolten AS, courtesy of the Norwegian Seafood Federation).

1.4 Sea lice and sea lice treatments

Sea lice

Sea lice (Figure 5) are species-specific naturally occurring seawater ectoparasites of the copepod family, favoring salmon hosts. This parasite is responsible for major economic losses for Norwegian fish farmers (Iversen et al., 2017). Infestations reduce the welfare of both farmed and wild salmonids and are a death threat to small wild salmonids migrating from rivers to the ocean feeding grounds in the spring/summer (Torrissen et al., 2013; Vollset et al., 2018). The life cycle of sea lice involves eight stages starting with Nauplii and finishing when the parasites become adult male or female (Hamre et al., 2013). The duration of each phase varies with sea temperatures and is expressed in degree-day^{-1} (Godwin et al., 2020). Sexually mature females can produce between 770 to 3190 eggs in their lifetime, with dispersal models suggesting robust survival and vast spreading potential of detached eggs through the currents of the upper sea layers (Eisenhauer et al., 2020).

Figure 5. Pre-adult and adult female sea lice. The egg strings are visible on the upper lice (Photograph: Thomas Bjørkan, the Norwegian Seafood Federation).

Treatment trends

During the last part of the 20th century up until 2014-2015, Norwegian salmon farmers relied primarily on chemicals to control sea lice. They applied different types of benzoylphenyl ureas, avermectins, pyrethroids, and organophosphates, as well as hydrogen peroxide through means of baths and medicinal feeds (Roth, 2000). The industry reliance on chemicals coupled with the production surge of 2000-2010 resulted in widespread resistances among sea lice (Aaen et al., 2015). Resistance to drugs forced farmers to look for alternative treatment methods, triggering an increase of interest in cleaner fish and the development of innovative mechanical treatments (Blanco Gonzalez & de Boer, 2017; Overton et al., 2018). Figure 6 shows that biological and mechanical treatment methods' development correlates with the peak and decrease of chemical use from 2014.

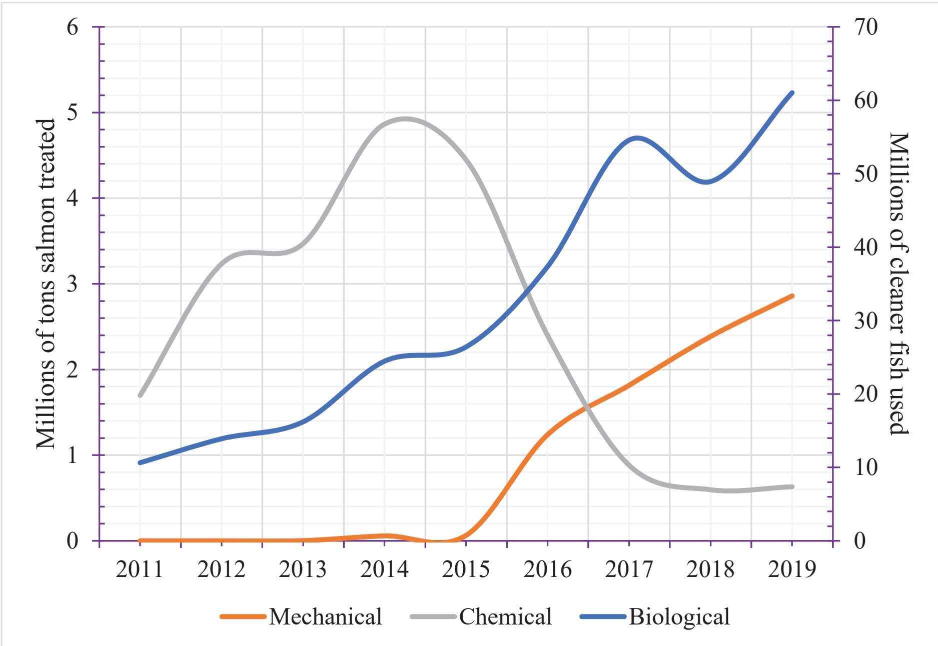

Figure 6. Evolution of sea lice treatments used by the Norwegian aquaculture industry between 2011 to 2019. Biological treatments are expressed in millions of cleaner fish used (right axis), while chemical and mechanical treatments are expressed in millions of tons salmon treated (left axis). Figure originally published in Paper III and reused in Paper V.

Biological treatments

Cleaner fish are used as biological treatment in the Norwegian aquaculture industry. Various types of cleaner fish are placed in the salmon net pens to eat lice directly off salmons (napping lice is more of a pastime for the cleaner fish and not his primary diet component). Salmon farmers have used this treatment method for decades, but it has gained momentum recently, and organized supply chains have emerged. All lumpfish (*Cyclopterus lumpus*) and a fraction of the Ballan wrasse (*Labrus bergylta*) are being farmed (Figure 7). Farmed lumpfish and wrasse are produced in land-based facilities, with an average rearing time of 6 and 18 months, respectively (Brooker et al., 2018). All species of Labridae (including Ballan wrasse) such as Goldsinny wrasse (*Ctenolabrus rupestris*), Corkwing wrasse (*Symphodus melops*), Rock cook (*Centrolabrus exoletus*), and to a lesser extent, Cuckoo wrasse (*Labrus mixtus*) are being fished in Norway and Sweden (BW, 2021b). Wrasse fishing takes place using traps in shallow water along the coast, starting near Gothenburg, moving along the Norwegian coastline until Nord-

Trøndelag. This activity is strictly regulated by the Norwegian Directorate of fisheries, with personal quotas attributed to each fisher and a limited fishing season during summer.

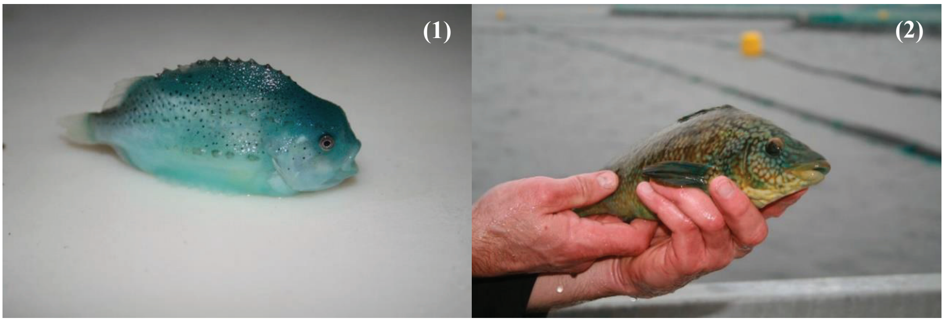

Figure 7. Examples of (1) lumpfish and (2) corkwing wrasse used by the Norwegian salmon farmers (Photographs: Øyvind André Haram, courtesy of the Norwegian Seafood Federation).

Mechanical treatments

The Norwegian aquaculture industry currently uses five mechanical treatment types (Table 1). Development of mechanical treatment units started in 2012-2013, followed by experimental treatments during 2013-2014. Large-scale commercial use rapidly increased from 2015, with most units built over the two following years.

Table 1. Mechanical treatment units currently used in Norway.

Type of treatment	Name of unit	Manufacturer
Thermal	Optilicer	Optimar
	Thermolicer	Scale AQ
Non-thermal	Hydrolicer	Smir
	FLS	Flatsetsund Engineering
	SkaMik	SkaMik

Thermal and non-thermal units consist of types of machinery using different operating principles. All systems are built on decks of either well-boats or service vessels, barges, and occasionally floating containers. The operation usually occurs with the unit positioned between two net pens; one contains the biomass being pumped into the system while the other receives treated fish flowing out of the delousing unit (Figure 8). Farmers systematically crowd the fish in the first net pen to pump the biomass through the system. Oxygen is added to improve fish welfare and ensure that concentrations never fall to dangerous levels for the fish.

Optilicer and Thermolicer are similar systems involving the immersion of fish for 20-30 seconds in seawater at temperatures up to 34°C. The difference of temperatures between the sea and the water in the system holds the delousing effect (Holan et al., 2017). It also means that the treatment temperature required increases and decreases in parallel with sea temperatures. Although the treatment affects both the salmon and the lice, the fish is less impacted due to its much greater mass (Grøntvedt et al., 2015; Roth, 2016). Non-thermal units used two main technologies: vacuum turbulence and spraying. Under treatment with Hydrolicer, the fish is pumped into/through a closed water pipe system and exposed to low negative pressure, creating vertical turbulences lifting sea-lice off the fish (Erikson et al., 2018). Both FLS and SkaMik spray the lice of the fish using low-pressure washers (Gismervik et al., 2017).

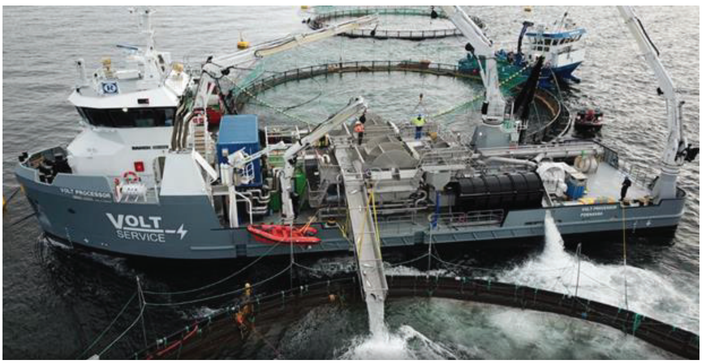

Figure 8. Service vessel equipped with Optilicer in operation between two net pens (Photograph: Per Håvard Fossheim, courtesy of Optimar).

Chemical treatments

The use of chemotherapeutant has diminished due to the increasing resistance of sea lice populations, but the Norwegian aquaculture industry still employed various delousing drugs in 2019 (NIPH, 2019). These chemicals are administered to the fish either through baths or as medicinal feed (Table 2). Baths involve crowding the fish in net pens using tarpaulin or onboard well-boats. Like for mechanical treatments, oxygen is added to ensure proper welfare/respiration of the fish in these unusually dense swimming conditions. On the other hand, medicinal feed does not require fish handling since the active substance is incorporated in feed pellets. Organophosphates like Azamethiphos and pyrethroids such as Deltamethrin affect the nervous system of sea lice by inhibiting acetylcholinesterase or binding to the nerves sodium channels. Both substances provoke nervous paralysis and the death of the parasite (Fallang et al., 2004; Sevatdal et al., 2005). Hydrogen peroxide's action is still ambiguous, but bubble formation in the lice hemolymph and gut lead to mechanical paralysis and detachment from hosts (Grant, 2002). Benzoylurea pesticides affect chitin synthesis, making them only effective during the development stages of sea lice (Macken et al., 2015). Finally, Emamectin benzoate binds to the glutamate-gated chloride channels of nerve cells, disrupting nerve impulses, resulting in paralysis and death (Stone et al., 1999).

Table 2. Chemotherapeutants currently used against sea lice in Norwegian net pens.

Administration	Active substance	Chemical category	Products
Bath	Azamethiphos	Organophosphate	Azasure, Salmosan
	Deltamethrin	Pyrethroid	Alphamax
	Hydrogen peroxide	Peroxide	Asperix, Nemona, Paramove
Medicinal feed	Teflubenzuron	Benzoylurea	Ektobann
	Diflubenzuron	Benzoylurea	Releeze
	Emamectin benzoate	Avermectins	Slice

1.5 Life cycle impacts

Life Cycle Assessments (LCA) have been applied to quantify the environmental impacts of salmon production over the past 15 years (Ellingsen & Aanondsen, 2006; Hognes et al., 2014; Papatryphon et al., 2004; Pelletier et al., 2009; Song et al., 2019; Winther et al., 2020). Results are concordant across studies: the composition of the feed and the type of production system are the two critical components affecting salmon's life cycle environmental impacts the most (Bohnes et al., 2018; Philis et al., 2019). Finding renewable feed ingredients with a low environmental footprint is one of the challenges the salmon aquaculture industry must tackle to improve its environmental profile (Aas et al., 2019; Silva et al., 2018). It is also critical to quantify and compare the environmental impacts of the different aquaculture systems in relation to critical parameters like FCR, mortality, local pollutions, and prevalence of diseases and parasites. Because most salmon LCA studies are case-specific and rarely reproducible, fairly comparing aquaculture systems remains challenging (Bohnes et al., 2018; Philis et al., 2019).

One aspect hindering equitable comparison between net pens and other aquaculture systems is the lack of knowledge about the life cycle environmental impacts attributable to biological conditions in the cages. There is an abundance of research describing the prevalence and the effect of diseases and parasites on salmon (Grefsrud et al., 2019; Hjeltnes et al., 2019), yet the LCA framework is still lacking a consensual approach to account for such local ecological impacts (Bohnes & Laurent, 2018; Ford et al., 2012; Nyberg et al., 2021). How health conditions relate to life cycle impacts in animal production systems is not well documented in the LCA literature. Only a few studies cover this topic in agriculture (Hospido & Sonesson, 2005; Mostert et al., 2019; Williams et al., 2015). It is possible to better account for the impacts of diseases and parasites using the available LCA methodology by determining the main contributors of mortality and accounting for the impacts of treatments (Philis et al., 2019; Winther et al., 2020). While existing salmon LCA already structurally account for the production inefficiencies generated by mortality by relating their functional unit to the economic FCR of the system, the specific contribution of diseases, parasites, and their treatments remain unknown. Efforts to account for treatments, particularly against sea lice, have been deficient in salmon LCA, especially given the increased treatment activity and shift in treatment mix seen recently in the Norwegian aquaculture industry (Figure 6).

1.6 Structure and objectives

The overarching goal of my Ph.D. thesis is to bring new knowledge supporting the reduction of life cycle environmental impacts in the salmon aquaculture industry. Early on, my supervisors and I decided to focus on aquaculture production systems since it has received less attention than aquafeed ingredients in the field of LCA. It is also one of the critical components affecting salmon's life cycle impacts. My Ph.D. thesis primarily builds on the research performed in Papers I, II, and III, with Papers IV and V supporting the thesis discussion (Figure 9). In Paper I, I conducted a literature review of salmon LCA. In this first stage, my objective was to investigate:

- How was the LCA methodology applied and inventory built by practitioners? Could trends and issues be identified?

- Was it possible to statistically compare LCA results, using aquaculture systems as criteria, despite the variety of confounding factors between studies?

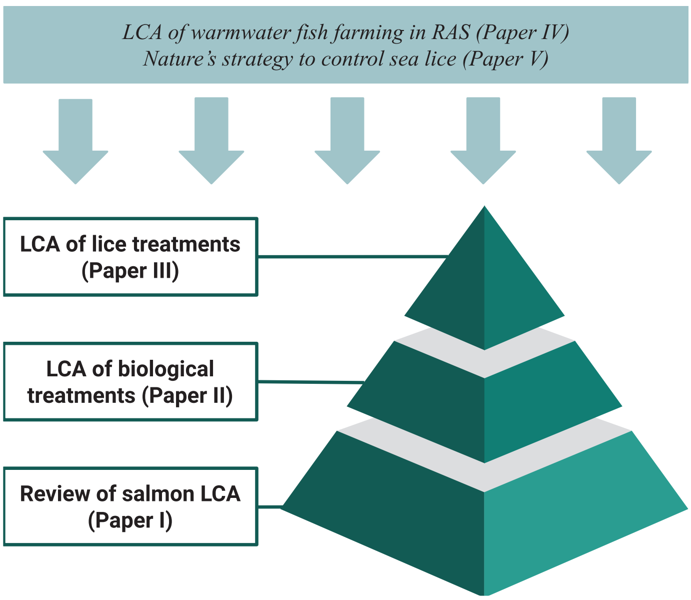

Figure 9. Illustration of the thesis structure. The core of the research was conducted in Papers I, II, and III. Papers IV and V bring important perspectives to the discussion.

The review made clear that the contribution of diseases and parasites was a knowledge gap skewing cross-study comparison of aquaculture systems involving net pens. It also showed me the difficulty of reusing LCA results since none of the LCA studied provided reusable inventory. Based on these findings and since I had access to high-quality statistics from Norwegian institutions, my supervisors and I decided to focus specifically on sea lice. This parasite was selected over viruses causing pancreas disease and infectious salmon anemia since it involves extensive treatment operations while viral diseases do not. In Paper II and III, I conducted original LCA studies to quantify the environmental impacts of the different sea lice treatments used by the Norwegian aquaculture industry. In this second stage, my goal was to study:

- How much life cycle impacts are currently generated by the biological, mechanical, and chemical treatments used by Norwegian salmon farmers?

- What is the contribution of the lice treatment mix and the treatment-induced mortality to the farmgate[2] salmon footprint?

- Is it possible to publish open-access, reusable LCA data while respecting the constraints from confidentiality agreements established with companies?

Finally, I further use the example of tilapia and clarias produced on land in a Swedish RAS facility (Paper IV) to discuss if land-based production systems could be one of the solutions to the Norwegian sea lice problem and a sustainable alternative for the industry. I expand further this discussion suggesting how using nature's strategy against sea lice in sea-based systems (Paper V) could represent a viable alternative enabling sustainable growth of the Norwegian aquaculture industry.

[2] In this context the term "farmgate" refers to the system boundaries used to characterize the environmental impacts. Farmgate means that the LCA stops at the gate of the salmon farm with fresh live salmon is ready for slaughter.

2 Material and methods

2.1 Life cycle assessment

LCA started to emerge as an environmental accounting tool for product comparison between the years 1970 to 1990 (Guinée et al., 2011). Today, LCA is a widely used method defined in ISO standards and ILCD handbooks (ILCD, 2010c; ISO, 2006a). In environmental management, it is the reference tool used to evaluate the environmental impacts associated with the production, use, and end-of-life treatment of goods and services. This framework is also useful for conducting consistent comparisons of anthropogenic production systems and alternative technologies according to their environmental impacts (Guinée, 2002). The LCA methodology is articulated around four main stages: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation. Since LCA is an iterative method, it means that each stage can be subject to revisions.

Goal and scope

Defining the goal and scope of the assessment is a critical step. It is where the practitioner defines the functional unit and the system boundaries of the system in consistency with the aim of the study. The functional unit must be clearly defined, measurable, and reflect the function of the product or service studied (Finnveden et al., 2009; ISO, 2006b). The choice of system boundaries marks the separation between the technosphere and the system assessed. It determines which processes are included and excluded from the system. Cut-off criteria can be understood as conditions for omissions of processes or flows that should have otherwise been within the boundaries. However, decisions to omit specific parts of the system are only allowed if the study's conclusion remains unchanged (ISO, 2006b).

Inventory analysis

During the inventory analysis, LCA practitioners collect data from various stakeholders and LCA databases. This phase is the most time-consuming task of the LCA. In practice, it consists of gathering qualitative and quantitative data of biophysical flows (material and energy inputs and emissions to air, water, and soil) involved in the processes modeled. Linear matrix algebra is used to convert process data into an inventory of substances, which are then further converted into environmental impacts using characterization methods. In LCA, there are two main inventory modeling philosophies: the attributional and consequential frameworks. Attributional LCA is often referred to as the descriptive/book-keeping method. It portrays actual or forecast value chain using historical, fact-based data, while consequential LCA is often depicted as a change-oriented and market-based framework evolving in a dynamic technosphere (ILCD, 2010c).

Impact assessment

During the impact assessment phase, the inventory of substances (in matrix form) is converted into impact categories scores using characterization methods. This conversion relies on characterization factors specific to each substance or group of substances. The choice of impact categories and characterization methods are interconnected and an important part of the impact assessment stage (ILCD, 2010a). Midpoint characterizations convert inventory of substances into indicators of emissions/resource consumption (e.g., ozone depletion, expressed in kg CFC-11 eq) while endpoint methods have a damage approach, aggregating environmental impacts into areas of protections (e.g., damage to ecosystems, expressed in species × year) (Huijbregts et al., 2016; ILCD, 2010b). Normalization and weighting are the two optional steps of the impact assessment. Normalization consists of relating the different impact category scores to a reference impact to facilitate comparisons, while weighting entails aggregating the normalized scores into a single impact using weighting factors (ILCD, 2010a).

Interpretation

Interpretation is often presented as the fourth and last phase of an LCA, but it is also an iterative process used every step along the way. Once environmental impacts are calculated, a contribution analysis is usually conducted by LCA practitioners to identify which processes and flows contributed the most and the least to the different impact categories. Sensitivity analysis is another interpretation tool applied to test how a change in a single parameter affects the results (scenario analysis usually tests several parameters at once). Finally, in recent years, methods to quantify the uncertainty of LCA modeling became more common. Most practitioners only quantify the uncertainty from data inventory (also called parameter uncertainty) using Monte Carlo simulations (Huijbregts et al., 2003).

2.2 Modeling choices

Throughout this Ph.D. thesis, I directly worked with and applied the LCA methodology. Together with the literature review of salmonids LCA, the assessments of lice treatments I have conducted constitute the core of this Ph.D. work (Paper I, II, and III). All my LCA work has been done in SimaPro and Excel.

Goal and scope

In the two original LCA I conducted, I used two types of functional units, referred hereafter as tier 1 and tier 2, both following cradle-to-grave system boundaries. Tier 1 functional units express impacts per cleaner fish (paper II) or per ton of salmon treated (paper III). Tier 1 units rely on value-chain data, primarily collected from companies. While these units provided valuable results, particularly to identify which processes and elementary flows contributed the most to the impacts of each value-chains (contribution analysis), tier 1 units could not provide impacts representative of the Norwegian sea lice treatment mix. It was not possible to aggregate the tier 1 impacts from both studies to determine the impact contribution of lice treatments to Norwegian farmgate salmon produced in net pens. Tier 2 functional units resolve this shortcoming by building on tier 1 units combining treatment and biomass statistical data of a representative group of Norwegian localities. This data allowed me to put in relation the quantities of cleaner fish used and of salmon treated (including the type of treatment applied) with the biomass output of the localities over their respective production cycles. This operation converted all tier 1 functional units into tier 2, expressing the environmental impacts of biological, mechanical, and chemical treatments per ton of salmon produced.

Inventory analysis

I applied the attributional LCA framework in both lice treatment papers. Allocations were required to divide the impacts of the fisher's boats and the associated antifouling paint required to model the impact of wrasse fisheries (Paper II) as well as for splitting the impacts of the different mechanical treatment units (Paper III). In each case, I used mass allocation. For the fisher's boats, I assumed a lifespan of 30 years and extrapolated the average biomass landed over the 2012-2018 period, while for the antifouling paint, I supposed a yearly renewal and used the 2018 landings. Impacts generated by the construction of mechanical units were allocated using a similar approach. I assumed eight years of operation time and combined treatment records, biomass statistics, and the number of units in use over time to derive the average treatment rates of each unit. The average treatment rate over 2017-2019 was extrapolated to the units' lifespan. I documented the use of cut-off criteria in Figure 1 of Paper II and Figure 2 of Paper III. The choice to include or exclude processes was made based on their perceived importance in the value-chain assessed, the data quality and availability, and eventually precedents from the literature. For instance, in Paper II, the exclusion of the construction of the reconverted cleaner fish production plants was supported by the work of Bergman et al. (2020).

Impact assessment

The selection of characterization method and impact categories was a delicate matter. It was important for me to stay consistent with the standard practices in the salmonid LCA literature but also apply the best method available. My choice was directly influenced by the scope of my work on lice treatments, the results of my review of salmonid LCA (Paper I), some LCA methodological articles (e.g., Steinmann et al. 2016), and my perception of the best characterization methods available. To determine the contribution of lice treatments to Norwegian farmgate salmon, it was clear that I had to work with midpoint impacts. LCA practitioners usually selected CML-IA, Cumulative Energy Demand (CED), and to a lesser extent ReCiPe (at the midpoint level) for their calculations. I opted for ReCiPe 2016 because many of the characterization factors used in this method are building and improving from those still used in CML-IA (Huijbregts et al., 2016; Van Oers, 2016). For comparison purposes, I used the same set of six impact categories in papers II and III: Climate Change (CC), Marine Eutrophication (MEU), Marine Ecotoxicity (MET), Land Use (LU), Water Use (WU), and CED. I used a limited number of categories to avoid diluting the results in too much information. Acidification was excluded because of the close correlation with CC impacts observed in the salmonid review. WU was originally calculated with ReCiPe but changed for AWARE during the review process of Paper II (Boulay et al., 2018). AWARE provided a more realistic characterization of water flowing through hydropower turbines than ReCiPe. This was particularly relevant for processes involving electricity consumption since the Norwegian mix is almost exclusively hydropower. While ReCiPe and AWARE indeed are more sophisticated methods, using them reduces the direct interoperability of lice treatment impacts I obtained with the farmgate salmon impacts available in the literature. Finally, normalization and weighting were not found necessary in my assessments.

Interpretation

In both lice treatment papers, interpretation was articulated around (1) a contribution analysis and (2) relating the impacts of the combined treatments to the impacts of farmgate salmon. Point 1 involved measuring the intrinsic intensities of the treatment value chains. In contrast, point 2 takes the results at a higher level, aggregating treatments together to reflect the impact generated by the Norwegian lice treatment mix. To support interpretations, I measured some uncertainty and sensitivity of the cleaner fish model (Paper II) and some of the uncertainty of the mechanical and chemical treatments (Paper III). I quantified the uncertainty from inventory data of the foreground system (collected first-hand) and the background system (from LCA databases) in both papers. Since I aimed to model national average of treatments, I tried to obtain multiple sources per process to account for production disparities (e.g., data of five lumpfish farmers was collected and merged to model this production process). For most processes, I measured the variability in the foreground system using triangular functions, including a minimum, a maximum, and a weighted average value for each input and output flows. The uncertainty was then calculated in SimaPro using Monte Carlo simulations. My supervisors and I considered that 1,000 iterations with a 95% confidence interval were robust enough considering the number of input parameters in each model (Heijungs, 2020). The sensitivity and scenario analyses conducted in Paper II covered both tier 1 and tier 2 functional units. They tested the sensitivity/reactivity of the results to (1) a change from Norwegian electricity mix to a European one, (2) two uneven sources reporting the number of cleaner fish deployed in Norwegian net pens, and (3) a scenario under which Norwegian authorities would ban wrasse fishing. Unfortunately, Paper III lacks sensitivity and scenario analyses due to its extended scope covering both mechanical and chemical treatments and the cumulated lice treatment burden (building on top of Paper II).

2.3 Data collection

Modeling biological, mechanical, and chemical lice treatments required extensive data collection, mainly production data from companies and statistics from Norwegian institutions. As a result, the bulk of the value-chains modeling is based on company data, while statistics helped perform allocations, distributions and support some assumptions. National treatment and biomass statistics are also the fundamental data used to convert tier 1 to tier 2 functional units for all treatments, using 307 Norwegian localities, representative of the salmonid production in the country.

Biological treatments - Tier 1 data (Paper II)

The production stage was based on the data of six cleaner fish farmers, 66 wrasse fishers, and including the feed recipes of three feed manufacturers. I used questionnaires to collect the farming/feed data for the accounting year 2017. While this approach facilitated the companies' reporting, it increased the possibility of a mismatch between inputs and outputs. This was particularly an issue for the wrasse farming value-chain since the production process lasted 18 months, and the farmer significantly increased his production between two generations. To reduce the risk of error, I used a weighted average for years 2017-2018 and checked that the input of feed, the output of fish, and the mortality level were in phase with the economic FCR reported by the farmer.

I conducted phone interviews with wrasse fishers, focusing on their last fishing season, which took place in the summer of 2018. The boat and antifouling paint allocation of wrasse fishing were modeled using statistics from the Norwegian Directorate of Fisheries during the 2012-2018 period. These statistics specific to the wrasse fishery included data about the fishers, their vessels, and landings of wrasse and other captured species (NDF, 2019). I modeled distribution using the data of one distributor, the sales reports of the farmers, and treatment statistics. My supervisors and I estimated that feed was the only critical input to the use process and opted to collect this data using a simple top-down approach based on the yearly sales of two leading cleaner fish feed manufacturers in Norway. This mix of process data and statistics allowed me to calculate the life cycle impacts of the different value chains per ton of cleaner fish produced, distributed, and used. For more details, see Figure 1 and Table 1 in Paper II.

Mechanical treatments – Tier 1 data (Paper III)

I collected data from five manufacturers using questionnaires to model the construction of the mechanical treatment units. The companies provided average data per unit produced based on 2018 technology. It was important to determine the number of treatment lines per unit, which varied between unit types/manufacturers. For instance, the number of lines per unit influenced how I divided equipment shared by several lines (e.g., oil boiler). Distribution was merged to production due to the simplicity and estimated low impact of this process. I appraised the distribution impacts based on the fuel consumed to sail from the treatment vessel homeport to the manufacturer's facility. To allocate the construction of mechanical units per ton of salmon treated, I used treatment and biomass statistics from BarentsWatch and the Norwegian Directorate of Fisheries over the years 2012-2019 (BW, 2021b; NDF, 2020a). Since the BarentsWatch data lacks details about mechanical treatments, I added the records of the VetReg database. Many farmers specify the type of mechanical treatments used in this database managed by the Norwegian Food Safety Authorities (NFSA, 2021b). These statistics, coupled with the number and production year of the units, gave me the possibility to derive each unit's use rate and perform the allocation.

Unlike for biological treatments, modeling the use phase was data intensive. I collected data from three fish farmers using Excel tables with records of different treatments performed in the farmers' localities between 2017 and mid-2019. The number of samples ranged from 12 to 30, depending on the treatment types. Samples for SkaMik lacked and were replaced by an average inventory of Hydrolicer and FLS (the two other non-thermal treatments). My fuel consumption calculations rely directly on the vessel data provided by the farmers. I also used the AIS data available on BarentsWatch to determine the location and sailing distances of the well-boats (BW, 2021a). This cumulated data allowed me to quantify the impacts of mechanical treatments per ton of salmon treated. Further details are available in Figure 2 and Table 2 of Paper III.

Chemical treatments – Tier 1 (Paper III)

I had difficulties accessing the production data of companies manufacturing chemical delousing products. Due to a lack of reply despite sustained data collection efforts, I modeled these processes using summaries of product characteristics published by the

Norwegian Medicines Agency (NMA, 2020). Distribution was also aggregated to production in this model. The import of delousing products to Oslo was calculated using open data of companies holding marketing authorizations and assumptions about their logistics. Distribution from Oslo to all aquaculture localities was simplified by using six logistic proxy hubs along the Norwegian coast: Stavanger, Bergen, Ålesund, Trondheim, Bodø, and Tromsø. The BarentsWatch treatment data for each active substance was added to derive the distribution shares of the different hubs (BW, 2021b). To aggregate products using the same active substance, I calculated market shares according to the sales statistics from the Norwegian Institute of Public Health (NIPH, 2020).

Overall, the same approach was applied to model the use phase of mechanical and chemical treatments. Here, only two out of the three farmers reported using chemical baths, providing a number of samples comprised between 10 and 13, depending on the active substance. Together with my supervisors, I estimated that only the active substance in medicinal feeds should be allocated to the treatments. In practice, it means that the production and distribution of chemical feed are already accounted for in the salmon life cycle since it directly substitutes a conventional feed, except for the active substance. I used a top-down approach to model medicinal feed treatments because the Ektobann, Releeze, and Slice data of the three farmers were scarce and uncertain. The Norwegian Food Safety Authority gave me access to their statistics, including the biomass treated in 2017, 2018, and 2019 (NFSA, 2021a). I combined this data with the total use of active substances in feed medicine over the same period (NIPH, 2019). I calculated the life cycle impacts of chemical treatments with this inventory, expressed per ton of salmon treated. Further details are available in Figure 2 and Table 2 of Paper III.

Functional unit conversion – Tier 2 data (Paper II and III)

To convert the tier 1 functional units of biological, mechanical, and chemical treatments into tier 2, I directly relied on the treatment data from BarentsWatch and the biomass statistics of the Norwegian Directorate of Fisheries (BW, 2021b; NDF, 2020a). Combining these datasets is necessary to link the number of treatments performed to the biomass in the net pens of each locality. Figure 6 (also displayed as Figure 1 in Papers III and V) illustrating the evolution of lice treatments is also based on these datasets. While the BarentsWatch data is openly downloadable online, biomass data at the locality level is considered sensitive by the Norwegian Directorate of Fisheries and can only be accessed by Norwegian research institutions using the Directorate's data for research applications. I selected a pool of 307 localities with a complete production cycle between January 2017 and June 2019 to correctly link treatments to their respective production cycles and obtain treatment impacts representative of the Norwegian lice treatment mix. This step is compulsory to obtain lice treatment impacts directly comparable to the impacts of farmgate salmon produced in net pens.

To avoid some outliers and confounding factors, I estimated an entire cycle to be contained between 52 and 104 weeks of continuous production at sea. These localities cover the 13 Norwegian production zones, the spring and fall salmon deployments in the sea, and the production of both Atlantic salmon and Rainbow trout. I excluded localities producing broodstock because these fish used for reproduction grow larger than market salmon. Their production can benefit from higher lice thresholds that can result in reduced treatments. I matched the treatment and biomass statistics in Excel using locality, week, and month numbers for my work in Paper II and III. I directly related the number of cleaner fish used to the quantity of salmon slaughtered to obtain the tier 2 functional unit for biological treatments.

It was unnecessary to link the number of cleaner fish to the salmon biomass being treated since the tier 1 functional unit was expressed per ton cleaner fish produced/fished, distributed, and used (and not per ton of salmon treated). On the other hand, for mechanical and chemical treatments, expressing the impacts per ton of salmon treated was the only obvious tier 1 functional unit alternative. Consequently, I directly linked the treatments to the biomass stock in the net pen being treated. When only a fraction of the locality stock was treated (quantities were not specified), I estimated the number of salmon being treated using the data provided by the three salmon farmers. Finally, the tier 1 functional unit was converted to tier 2 by combining the intensities of the different mechanical and chemical treatment value chains (expressed per ton of salmon treated) with their usage frequency calculated over the 307 production cycles studied.

3 Results and discussion

3.1 Life cycle impacts of production systems (Paper I)

One of the salmon LCA review objectives was to compare the impacts of salmon produced in different aquaculture systems. I identified 4-clusters of systems: (1) closed, sea-based, (2) open, land-based, (3) open, sea-based, and (4) closed, land-based. Life cycle impacts and the economic FCR were sourced from the 24 original salmon LCA reviewed, using 1 ton of farmgate salmon as standard functional unit. Closed, sea-based structures consisted of innovative systems primarily used for research and development, such as a solid wall system and a marine floating bag. Category 2, 3, and 4 included FTS, net pens, and RAS, respectively. To compare life cycle impacts, I selected the four most prevalent impact categories used in the literature: global warming potential (equivalent to CC), acidification potential, eutrophication potential (not equivalent to MEU), and CED.

Descriptive comparison

Average results per system clusters per impact category show notable differences (Table 3).

Table 3. Average life cycle impacts of different types of aquaculture systems assessed in the salmon LCA literature according to the four most used LCA impact categories.

	CC (kg CO_2 eq)	AP (kg SO_2 eq)	EP (PO_4 eq)	CED (MJ eq)
(1) closed, sea-based	2,404	15.1	26.7	54,620
(2) open, land-based	2,613	16.3	50.6	75,943
(3) open, sea-based	2,933	18.7	47.3	37,913
(4) closed, land-based	6,414	26.7	17.3	133,220

CC and AP impacts for categories 1-3 are concentrated in narrow ranges evolving between 2,404-2,933 kg CO_2 eq and 15.1-18.7 kg SO_2 eq. RAS impacts are significantly higher, scoring 6,414 kg CO_2 eq and 26.7 kg SO_2 eq for these two categories. For CED, the land-based systems display significantly higher energy requirements (75,943-133,220 MJ eq) than sea-based systems (37,913-54,620 MJ eq). Somehow similar trends between CC, AP, and CED are not surprising since energy consumption, carbon emissions, and acidification correlates partially or totally, depending on how electricity is produced. RAS are known to have higher impacts in these categories due to their fundamentally higher energy demand to pump, recirculate, and treat the water in the system. For EP, the situation is inverted, with the closed systems displaying the lowest impacts (17.3-26.7 kg PO_4 eq) and the open systems the largest ones (47.3-50.6 kg PO_4 eq). This advantage can directly be attributed to the collection and treatment of effluents made possible by closed systems but not by open ones.

Statistical comparison

A descriptive comparison of averaged aquaculture systems impacts comes with severe limitations from multiple confounding factors and limited sample sizes. Confounding factors can be found at every stage of each LCA in both inventory data and methodological choices. They include critical aspects ranging from the location, production year, type of feed, FCR to the selection of LCA databases and characterization methods. I tried to address this issue in the review by statistically testing the impacts of the different clusters using a single parametric statistical protocol. The objective was to test if a simple non-discriminant approach could compare the clusters' impacts despite the confounding factors and limited data samples.

The Life Cycle Impact Assessment (LCIA) results and the FCR of the salmon LCA reviewed were statistically tested following three different grouping protocols (see Figure 1 in Paper I). First, according to the 4-clusters of aquaculture systems mentioned above. Second, by aggregating the data into 2-clusters of closed and open systems. Third, by compiling the data into either land-based or sea-based systems. The subsequential dual grouping allowed to increase sample sizes compared to the 4-clusters analysis. I applied the ANOVA and Welch t-tests to compare the means of the different clusters, depending on if the clusters tested had homogenous or heterogenous variances. ANOVA and Welch allowed me to determine if significant statistical differences between the clusters means existed or not. When significant differences were found, the Games-Howell post hoc test was applied to determine which clusters within the group generated the differences. One of the reviewers criticized using two statistical methods and underlined the importance of using a single non-parametric method for the whole study. Unfortunately, this approach was not possible based on the data currently available in the literature. The Kruskal-

Wallis statistical test, the non-parametric equivalent to ANOVA, was ineligible due to heterogenous distributions.

Across the different groups, FCR data validated the assumption of normality, and I was able to apply this statistical protocol. On the other hand, a large fraction of the LCIA data failed the Shapiro-Wilk normality test and was consequently excluded of the analysis. For the 4-clusters comparison, this includes CC and AP (50% of the data). For the two groups comparing open vs. close and sea-based vs. land-based clusters, 75 and 100% of the LCIA data failed the normality test and was left out from further testing. One of the assessors pointed out during my mid-term Ph.D. defense that it is generally assumed within the LCA community that LCIA data is log-normal distributed. It could be valuable to test if the non-normal clusters validate this assumption.

Despite the reduced statistical comparability, a portion of the data tested with ANOVA/Welch and Games-Howell demonstrated some significant statistical differences between the LCIA and FCR data of clusters. The following conclusions could be drawn from the statistical analysis:

- Sea-based systems require significantly less energy than land-based systems (41,768 vs. 96,771 MJ eq)
- Land-based systems have a significantly lower FCR than sea-based systems (1.12 vs. 1.26)
- Closed systems likely have a significantly lower eutrophying potential than open systems (20.5 vs. 48.8 kg PO4 eq)

3.2 Impact contribution of lice treatments to net pen salmon (Paper II and III)

Total lice treatments

Results of Paper III integrate and build on results of Paper II, providing the total life cycle impacts of lice treatments for CC, MEU, MET, LU, WU, and CED per ton of salmon produced (Table 4). Mechanical treatments dominate the impacts of the Norwegian lice treatment mix due, by far, to the highest usage frequency and medium impact intensities from its value-chains. Chemical treatments add significant impacts to the mix, primarily because of the very high impact intensity linked to hydrogen peroxide baths. Biological treatments add low impacts to the mix across impact categories. Overall, lice treatment impacts are driven by: (1) increased mortality, (2) fuel use, particularly from large well-boats, (3) production of hydrogen peroxide, and (4) construction of mechanical treatment units. Despite high intrinsic MET, baths and medicinal feed chemicals generate low toxic impacts per ton of salmon produced. The MET emissions are driven by the metals required to produce mechanical units. On average, increased mortality is the process dominating lice treatment impacts, particularly for MEU, LU, and WU (Table 4). It is essential to highlight that directly adding the total lice treatment impacts with the increased mortality to the impacts of farmgate salmon will involve double counting. LCA already captures the inefficiency generated by the increased mortality from lice treatments through the accountancy of material and energy required to produce a standard quantity of salmon.

Table 4. Impacts of the different types of sea lice treatments, including the specific contribution of mortality, per ton of salmon produced in Norway between 2017 to mid-2019. The contribution of increased mortality to total lice treatments is displayed in gray (in percentage).

Impact category	Mechanical treatments	Chemical treatments	Biological treatments	Total lice treatments	Contribution mortality
CC (kg CO2 eq)	73.1	52.8	8.0	**133.9**	37%
MEU (kg N eq)	0.72	0.07	0.10	**0.88**	89%
MET (kg 1,4-DCB eq)	15.2	2.4	0.2	**17.7**	14%
LU (m2a crop eq)	37.1	4.2	1.2	**42.6**	95%
WU (m3)	54.9	13.5	9.0	**77.4**	76%
CED (MJ eq)	957	1,061	177	**2,195**	27%

Contribution to net pen salmon

Relating total lice treatments to the review results for all categories was hindered by using different characterization methods and impact categories. Baseline net pen farmgate salmon for CC and CED were sourced from the Paper I (open, sea-based systems), while those for MEU, MET, LU, and WU were derived from single studies coupled to assumptions. For more details, see section 4.2 of Paper III. Overall, the contribution of total lice treatments to salmon net pen is significant, particularly for CC, MET, and CED

(Table 5). However, because of the large influence of treatment-induced mortality, which was already accounted for in the literature, absolute values only slightly increased from their baselines (especially MEU, LU, and LU). For CC and CED, the average impacts of open, sea-based systems increased from 2,933 to 3,017 kg CO2 eq and from 37,913 to 39,510 MJ eq. This means CC increased by 2.86% and CED by 4.21% from baseline. While this is a significant increase that needs to be accounted for to make a fairer comparison of aquaculture production systems impacts involving net pens, such augmentation will likely not affect the descriptive and statistical comparison results performed in Paper I.

Table 5. Impacts of net pen salmon with and without total lice treatments. Increased mortality was subtracted from original impacts to avoid double counting. The percentage contribution of total lice treatments to net pen salmon is displayed in gray.

Impact category	Net pen Salmon without increased mortality	Net pen Salmon with total lice treatments	Contribution total lice treatments
CC (kg CO2 eq)	2,883	**3,017**	4.4%
MEU (kg N eq)	75.8	**76.7**	1.1%
MET (kg 1,4-DCB eq)	252	**270**	6.6%
LU (m2a crop eq)	3,902	**3,944**	1.1%
WU (m3)	5,665	**5,743**	1.3%
CED (MJ eq)	37,314	**39,510**	5.6%

3.3 LCA methodology: data reusability (Paper I, II, and III)

One of the main difficulties I encountered performing the cross-study analysis of LCIA results in Paper I was the lack of transparency and reproducibility in the salmon LCA literature. LCA is an adaptive tool requiring flexibility to allow practitioners to tailor the method to the product/system assessed. This variability in methodology and data generates confounding factors making comparisons more challenging. Having access to reusable LCA models would give room and flexibility to correct some of the most critical confounding factors before performing comparisons. In practice, this could mean recalculating the model using a different characterization method or changing the feed to standardize that aspect of the production. While LCA practitioners usually describe their main methodological choices (e.g., allocation method) and disclosed aggregated foreground inventory, none of them made their model available for reuse. Without a detailed inventory, including processes from LCA databases, it is impossible to duplicate the assessment. The confidentiality often required by companies and other data owners can partially explain this lack of transparency, but it cannot explain why this practice is systemic in the literature.

In the LCA conducted in Paper II and III, I tried to demonstrate that it was possible to disclose inventories despite the limitations imposed by confidentiality agreements. Modeling the environmental impacts of lice treatments at the national level facilitated my transparency efforts. In many cases, working at this scale allowed me to gather several data sources for the same process. The data from the different sources were anonymized and aggregated into a single process openly publishable. Submitting Paper II and III to the Journal of Industrial Ecology encouraged and facilitated disclosure of my inventories. It was encouraged by the data openness badges schemes used by the journal promoting studies providing reusable data. It was facilitated by the publication of the inventories on the open access repository Zenodo. Paper II received the gold-gold open access badge with inventories available in .xlsx and .CSV formats, making them both human and machine-readable. Paper III is under review at the time of writing. Inventories of Paper III will be published following the same open access procedure. Overall, this means the LCA practitioners can either manually recreate the processes in the model using the data available in Excel or import the .CSV files into SimaPro and access the system processes they contain. I exported the biological lice treatment processes in "system" format for simplicity. Reusability and convenience could be improved further by exporting them as unit processes instead.

3.4 LCA methodology: limitations (Paper I, II, III)

Throughout my Ph.D. work, I have been challenged by the limitations of the LCA methodology on several occasions. It is important to identify and comprehend these limitations to interpret the comparison of life cycle environmental impacts performed in Paper I and the quantitative assessments of lice treatments performed in Paper II and III.

Incomplete environmental impacts

Overall, LCA is internationally recognized as a reference assessment method well suited to quantify some global and regional environmental impacts generated by anthropogenic value chains (Finnveden et al., 2009; ISO, 2006a). Despite certain qualities, LCA still lacks a consensual framework to effectively account for local ecological impacts, particularly regarding the loss of biodiversity and ecosystems, including the critical services they provide (Chaplin-Kramer et al., 2017; Curran et al., 2011; Liu & Bakshi, 2019). Endpoint impact categories available in methods like ReCiPe and LC-impact provide characterization factors to quantify the life cycle impacts of product and services on ecosystem quality (Huijbregts et al., 2016; Verones et al., 2020). However, Paper I show that midpoints impact categories are almost systematically selected over endpoints by practitioners. Adding a characterization step to evaluate damage pathways to areas of protection adds significant complexity and uncertainty to the results. It is also often more intuitive for businesses to relate to midpoint categories like CC or WU to optimize their production processes (Bare et al., 2000). I would argue that a negative consequence of this LCA trend leads to overestimating the importance of highly publicized impacts like CC compared to impacts more complex to apprehend, like ecosystem quality. Extensive research has also been taking place to incorporate marine ecological impacts into the LCA framework (Woods et al., 2016), including new attempts to quantify the effects of marine plastics through projects like ATLANTIS (https://www.atlantis-erc.eu/) and MarILCA (https://marilca.org/). Still, little to no integration of aquaculture-specific impact categories took place since the suggestions of Ford and colleagues in 2012. These limitations have direct consequences on Paper I, II, and III results by making any attempt to make environmental impact comparisons between aquaculture systems incomplete. In practice, it means that impacts from land-based and sea-based systems on terrestrial and marine ecosystems quality are not directly weighted in the environmental tradeoff. This lack of information could skew the comparison and mislead stakeholders towards a particular system.

Animal welfare

While technically not an environmental impact, adding indicators to measure animal welfare to the LCA framework is actively discussed in the literature since it is a critical component of sustainability (Ford et al., 2012; Scherer et al., 2018; Sonesson et al., 2016). This is particularly significant for anthropogenic systems involving the production or the harvest of animals. In addition to being critical for ethical reasons, animal welfare has indirect effects on environmental impacts through key production parameters like FCR, diseases, parasites, and overall mortality (Hocquette et al., 2014). For all these reasons, fish welfare is a critical indicator that matters when comparing the environmental impacts of salmon aquaculture production systems. Identifying which aquaculture system could provide the optimal fish welfare conditions is challenging since different systems rely on different indicators (Kolarevic et al., 2018). Fish welfare is of concern in net pens mainly because of the different lice treatments used by farmers. Mechanical and chemical treatments involving fish handling generate a lasting cortisol response in the fish (Delfosse et al., 2021; Hoem & Tveten, 2020). Thermal treatments above 28°C are suspected of causing pain and injuries (Gismervik et al., 2019; Nilsson et al., 2019), which led the Norwegian Food Safety Authority to recommend the phase-out of treatments at these temperatures (NFSA, 2019). However, a recent press release suggests that Norwegian authorities turned around, leaving the salmon farmers to use thermal treatments despite the fish welfare concerns (IntraFish, 2021). Fish welfare is not only a concern for the salmon but also for cleaner fish. Biological treatments illustrate well the limitation of LCA on that aspect. From a strict life cycle perspective, the additional environmental impacts from cleaner fish are negligible (Paper II and III). On the other hand, from a fish welfare perspective, the current use of lumpfish and wrasse raises major ethical and sustainability questions. These concerns are broad, including the management of the stock of wild wrasse, geographic transfers of species, but also the fish living conditions in salmon net pens and their low survival rates.

3.5 Land-based production of salmon in Norway (Paper IV)

Historically, Norwegian land-based production systems for salmon have been limited to hatcheries and smolt production sites, with FTS used through the past decades and the emergence of RAS technology in more recent years (Dalsgaard et al., 2013; Martins et al., 2010). In response to the growing biological challenges in net pens affecting production costs and volumes, the development of new land-based facilities has been gaining traction in Norway and abroad (Lekang et al., 2016). Investments in RAS, FTS, and hybrid FTS have been increasing substantially, with an apparent interest among farmers in producing bigger smolts (between 0.2 to 1 kg) and taking the whole salmon production cycle on land (EY, 2019). The common sentiment in the literature suggests that using land-based production systems instead of open cage systems generates environmental trade-offs (Ayer & Tyedmers, 2009; Badiola et al., 2018; Liu et al., 2016; Song et al., 2019).

Positive trade-off

Land-based systems offer more control of the rearing environment with a clear physical separation between the fish and natural ecosystems (Bergheim et al., 2009). This allows farmers to control the common biological challenges from viral diseases (e.g., infectious salmon anemia and pancreas disease), parasites (sea lice and amoebae), and mitigate most escapes (Lekang et al., 2016). Lice treatments are consequently avoided altogether on land. RAS and hybrid FTS can also enable farmers to collect and treat effluents to various degrees, significantly reducing the direct emissions of effluents to marine and freshwater bodies (Meriac, 2019). However, this advantage is not specific to land-based facilities but a characteristic of closed systems capable of filtering a significant amount of feed and feces effluents and denitrify the water with biofilters (see 3.1). Reusing nitrogen and phosphorus-rich sludge can be achieved in freshwater conditions but is more challenging for salmon reared in marine or brackish waters because of high salt concentrations (da Borso et al., 2021). Moreover, the statistical results of paper I show that land-based systems have a significantly lower FCR than sea-based ones. This favorable tradeoff should be confirmed in the light of new salmon LCA studies available since Song et al. (2019) contradicts these findings with an economic FCR as high as 1.45. Overall, the positive tradeoffs described in the literature are in accordance with the life cycle impacts comparisons I performed in Paper I.

Negative trade-off

Producing on land significantly increase energy, water, and land requirements (Badiola et al., 2018; Hilmarsen et al., 2018). The degree of recirculation and water type (fresh, brackish, or marine) directly influence the demand for energy and water. As a rule of thumb, the proportion of water recirculation increases in parallel with the energy consumption (Bergheim et al., 2009). This is to satisfy the energy demand of pumps and water treatment systems, which are shown to dominate the electricity demand of salmon grow out in recirculating systems (Song et al., 2019). Water consumption is negatively correlated with the energy demand in this situation: the more recirculation, the lower are the make-up water requirements. This means that an FTS system will have significantly lower energy and substantially higher water footprints than a RAS system (Bergheim et al., 2009). The use of marine water increases energy demand in both FTS and RAS. The water must be pumped from the sea to the shore-based facility (freshwater sources are often higher than the facility, beneficiating from gravity). Overall, both FTS and RAS systems will have higher direct land-use impacts than net pen systems. Recently, Sintef estimated that moving the whole Norwegian salmon production into either hybrid FTS or RAS systems would require approximately 8.3 or 11.7 km2 of land, respectively (Hilmarsen et al., 2018). The descriptive and statistical comparisons of LCIA results from Paper I are in agreement with the negative trade-off described in the literature.

A word about energy

In Paper IV, my colleagues and I investigated the life cycle impacts of tilapia and clarias produced in a commercial RAS farm in Sweden. The main objective of this analysis was to quantify the environmental performances of indoor RAS warmwater farming taking place in a temperate climate and identify improvement potentials of the farm. A functional unit of 1 kg filleted tilapia and clarias was used to calculate freshwater eutrophication, CC, CED, and LU impacts. Interestingly, results showed that feed dominates the life cycle impacts associated with the production of tilapia and Clarias fillets across all impact categories, except for the energy demand of tilapia. This finding somehow nuances previous results that the energy consumption of RAS systems is systematically a driver of life cycle environmental impacts.

While encouraging for the development of RAS technology, these findings are not directly transferable to salmon RAS and only valid if a renewable-rich electricity mix is used. There are significant biological and technical differences between tilapia/clarias and salmon RAS, not necessarily favoring salmon (e.g., higher water quality requirements). So far, LCA of salmon reared in RAS have identified energy consumption as a major impact driver (Ayer & Tyedmers, 2009; Song et al., 2019; Wilfart et al., 2013). These results are in adequation with the significantly higher energy consumption of RAS found in Paper I. It is important to develop food production systems like RAS recognized for reducing pressure on marine ecosystems. However, it should not be done at the expense of total energy consumption. Moving the 1,300,000 tons of Norwegian salmon into RAS will require an additional 11.7 TWh to power salmon grow out compared to the current net pen production (Hilmarsen et al., 2018). It would represent a 7.6% increase of the 153 TWh produced in Norway in 2020 (NMPE, 2021). In my opinion, apprehending renewable electricity from hydropower, wind, and solar as abundant raises concerns. It underestimates the biodiversity/ecosystem impacts of renewable infrastructures (Gracey & Verones, 2016; Rehbein et al., 2020). It also ignores wind and solar equipment dependence on large quantities of metals and rare earths (Ali et al., 2017; Vidal et al., 2013) and the challenges linked to intermittent power generation (da Silva Lima et al., 2021; Perez-Arriaga & Batlle, 2012).

3.6 Controlling sea lice infestations to improve sea farming (Paper V)

Infestations from parasites and diseases are common in animal production systems characterized by large volumes of concentrated individuals (Mennerat et al., 2010; Thamsborg et al., 1999). In salmon aquaculture, the affluence of fish/hosts fuels the continuous production of sea lice year-round (Heuch & Mo, 2001). Sea lice are primarily a threat to wild salmon stocks, particularly for juvenile salmon migrating from Norwegian rivers through the fjords to reach the ocean's feeding grounds in the spring. In addition, lice treatments reduce production efficiency and degrade salmon welfare in net pens (Barrett et al., 2020). Many consider sea lice infestations to be the main bottleneck hindering further growth of the Norwegian aquaculture industry, particularly with the implementation of the traffic light system (Grefsrud et al., 2019; NMTIF, 2017).

In Paper V, my colleagues and I suggested adopting elements of nature's original coping mechanism to regulate the host/parasite balance between salmon and sea lice. Before the development of industrial farms in open cages, sea lice infestations were naturally controlled by the migration cycles of wild salmon (and exposure to fresh and brackish waters), decreasing the number of hosts in the fjords for long periods. In the winter when the reproduction factor of sea lice is at the lowest, most wild salmons are off the coast feeding or in the rivers. While sea trout could remain in the brackish water of the fjord in that period, no salmon were available for sea-lice reproduction and survival, considerably reducing hosts/parasites interactions.

It means that juvenile salmon could cross the fjords without encountering significant sea lice threats in the early spring. We argue that the current fallowing regulations could be improved by mimicking nature's strategy in the winter. The approach is simple: minimizing contacts between parasites and hosts for a sufficient time period and scale to bring down the concentration of sea lice in the fjords. The mandatory two months of fallowing is sufficient since lice can survive approximately 20 days in seawater at seven degrees in the winter (Samsing et al., 2016). However, in order for the winter fallowing to be effective, the distance between nearby farms and generation zones must increase substantially. This can be achieved by moving or slaughtering salmons in net pens or by keeping the fish in closed controlled systems (sea-based or land-based) for a sufficient period of time.

According to our suggestions, net pens would remain the main aquaculture system used for salmon grow out. If challenges from diseases and parasites can be significantly mitigated by mimicking the natural host/parasite balance, a substantial negative tradeoff of open sea-based systems could be alleviated. At the same time, Norwegian net pens conserve a key advantage over land-based production systems: excellent sea farming conditions without significant infrastructure and energy requirements. It does not mean that there is no place for alternative land-based and sea-based designs in our approach. Having strategic production capacities of RAS, FTS, and closed sea-based systems would be essential to keep production flexibility throughout the salmon life cycle. Increasing the RAS and FTS production capacities could allow farmers to keep their fish longer on land. Producing larger smolts could help regulate production time and transfers at sea under the coordinated winter fallowing of the generation zones we recommend. All hosts must be unavailable for sea lice in the area under confinement for the fallow to have an effect. While moving salmon generations to other parts of the production area is an alternative, keeping the biomass in closed sea-based systems can also give farmers additional flexibility.

4 Conclusions

Throughout this thesis, I have compared the life cycle environmental impacts of the main salmon aquaculture systems. I have also quantified the LCA impacts of the Norwegian lice treatment mix since sea lice is the main challenge currently hindering the growth of sea farming and driving the development of alternative land-based and sea-based systems. The Norwegian aquaculture industry is at a crossroads. I hope to provide results and suggestions that can help stakeholders towards environmentally sustainable salmon production.

- It is possible to make a simple statistical comparison of salmon LCIA results across studies despite confounding factors and small sample sizes. I statistically demonstrated that (1) sea-based systems require significantly less energy than land-based systems, (2) land-based systems have a significantly lower FCR than sea-based systems, and (3) closed systems likely have a significantly lower eutrophying potential than open systems.
- The Norwegian lice treatment mix adds significant impacts to the farmgate salmon environmental footprint, particularly to CC, CED, and MET. Overall, lice treatment impacts are driven by increased mortality, fuel use (particularly from large well-boats), production of hydrogen peroxide, and construction of mechanical treatment units. However, the environmental impacts of the treatment mix newly calculated are unlikely to affect the LCA results comparisons between net pens and other salmon aquaculture systems. The impact addition is too small, particularly since the mortality increase induced by treatments is likely already accounted for in the literature.
- Data reusability and interoperability are essential to improve LCIA results cross-study comparisons. Data aggregation and anonymization are helpful methods that can allow LCA practitioners to publish inventories despite confidentiality agreements. In Paper II and III, I illustrated how inventories could be shared in open access repository in both human and machine-readable formats.
- Comparing the performances of salmon aquaculture systems with LCA is not enough to capture the complete picture of environmental tradeoffs between systems. I argue that in its current state (and because of how it is applied), the

LCA framework underestimates the impact on ecosystem quality and fails to account for fish welfare.

- Finally, I suggest using elements of nature's strategy to control sea lice infestations recommended by Stene and colleagues to mitigate sea lice challenges, improve the environmental profile of open sea-based systems, and increase the sea-based production output. I argue in favor of a simpler low technology solutions like this, in opposition to the massive transfer of biomass into capital and technology-intensive systems such as land-based facilities.

5 Future outlook

Providing a comprehensive picture of the environmental tradeoffs of salmon aquaculture systems to stakeholders is a challenging exercise. Here are, I think, important aspects that should be addressed in the future to improve comparisons and offer better support to decision-makers:

- LCA of aquaculture production systems should be generally required, particularly for new systems receiving permits from the Norwegian Directorate of Fisheries.
- LCA of offshore production systems used in Norway should be conducted. Special attention should be given to measure the impacts of logistics because of increased sailing distances.
- Further research is needed to assess the capacity of closed sea-based systems to collect nitrogen and phosphorus emissions. For example, how would these perform compared to those of other systems?
- Further research is required to measure the freshwater and marine sludge recycling potential in Norway. What is the environmental profile of the marine sludge desalting process?
- Local eutrophication characterization factors should be developed for the Norwegian coastline to measure the eutrophying potential of the current emissions of effluents.
- Applying the LCA methodology in its current state is not enough to make a comprehensive comparison. Additional assessments of biodiversity, ecosystems, and fish welfare impacts are necessary.
- Stronger incentives for data reusability and interoperability are needed for cross-studies comparisons. Scientific journals have the possibility to enforce this practice through the publication process if they act in concert.

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Paper I: Literature review of salmonid aquaculture LCA

ERRATA

In section 3.3.1, "Overview of LCIA and FCR Results" of Paper I, the two citations used in the first sentence are wrong. The maximum GWP impact of close land-based systems of 13,622 kg CO2 eq was reported by Samuel-Fitwi et al. (2013), with reference number 50 and not d'Orbcastel et al. (2009) with reference number 53. In addition, the minimum GWP impact of close land-based systems of 1,157 kg CO2 eq was reported by d'Orbcastel et al. (2009), with reference number 53 and not Dekamin et al. (2015) with reference number 54.

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Review

Comparing Life Cycle Assessment (LCA) of Salmonid Aquaculture Production Systems: Status and Perspectives

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Abstract: Aquaculture is the fastest growing food sector worldwide, mostly driven by a steadily increasing protein demand. In response to growing ecological concerns, life cycle assessment (LCA) emerged as a key environmental tool to measure the impacts of various production systems, including aquaculture. In this review, we focused on farmed salmonids to perform an in-depth analysis, investigating methodologies and comparing results of LCA studies of this finfish family in relation to species and production technologies. Identifying the environmental strengths and weaknesses of salmonid production technologies is central to ensure that industrial actors and policymakers make informed choices to take the production of this important marine livestock to a more sustainable path. Three critical aspects of salmonid LCAs were studied based on 24 articles and reports: (1) Methodological application, (2) construction of inventories, and (3) comparison of production technologies across studies. Our first assessment provides an overview and compares important methodological choices. The second analysis maps the main foreground and background data sources, as well as the state of process inclusion and exclusion. In the third section, a first attempt to compare life cycle impact assessment (LCIA) and feed conversion ratio (FCR) data across production technologies was conducted using a single factor statistical protocol. Overall, findings suggested a lack of methodological completeness and reporting in the literature and demonstrated that inventories suffered from incomplete description and partial disclosure. Our attempt to compare LCA results across studies was challenging due to confounding factors and poor data availability, but useful as a first step in highlighting the importance of production technology for salmonids. In groups where the data was robust enough for statistical comparison, both differences and mean equalities were identified, allowing ranking of technology clusters based on their average scores. We statistically demonstrated that sea-based systems outperform land-based technology in terms of energy demand and that sea-based systems have a generally higher FCR than land-based ones. Cross-study analytics also strongly suggest that open systems generate on average more eutrophying emissions than closed designs. We further discuss how to overcome bottlenecks currently hampering such LCA meta-analysis. Arguments are made in favor of further developing cross-study LCA analysis, particularly by increasing the number of salmonid LCA available (to improve sample sizes) and by reforming in-depth LCA practices to enable full reproducibility and greater access to inventory data.

Keywords: LCA; life cycle assessment; environmental impacts; aquaculture; salmon; trout; production systems; review; cross-study comparison

1. Introduction

Population growth, dietary shifts and resource challenges in capture fisheries and agriculture are driving the development of aquaculture production systems worldwide [1]. By 2050, the global population is expected to reach 9.8 billion [2], and robust economic growth in developing nations is expected to lead to a dietary shift towards higher consumption of meat and dairy [3]. These commodities are known to carry a high environmental burden [4]. While one can hardly argue that social development is negative, population growth and higher affluence are expected to generate additional pressure on fragile ecosystems already severely threatened by a variety of adverse human impacts [5]. Current intensive and linear production systems of food, feed, fiber, and bioenergy are largely exceeding the earth's capacity to regenerate the consumed biomass, exposing our societies to serious environmental and food security risks in the long run [6]. Global food production cannot continue to depend on agriculture and capture fisheries to the same degree in the future. Fisheries have been stagnating since the 1990s, constrained by the natural production limits of wild populations [7]. Meanwhile, agricultural systems are facing various challenges including decreasing soil fertility and arable land availability [8,9].

Aquaculture is one of the most efficient food production systems the world will rely upon to meet tomorrows demand for animal-sourced foods and reduce current environmental impacts associated with the supply of animal protein [1,10]. Since the 1980s, one of the largest developments in aquaculture has been driven by carnivorous salmonids, especially Atlantic salmon (*Salmo salar*), rainbow trout (*Oncorhynchus mykiss*), and coho salmon (*Oncorhynchus kisutch*). Salmonid aquaculture is central to the global seafood landscape since it alone accounts for 18.1% of the value and 7.4% of the biomass of the world trade in fish and fish products [1]. This production is especially important in Northern Europe (Norway, Ireland, Scotland, Faroe Islands, Iceland) and North and South America (Chile andCanada), in countries with abundant access to cold/temperate marine waters and extensive coastlines. Salmonids are primarily consumed in the United States, Europe, and East Asia. While consumption growth slows down in these traditional markets (but remain positive), consumption is now increasing the most in emerging markets such as Southeast Asia and Latin America [11]. Salmonid species are particularly appreciated for their taste, reddish-orange colored fillets, high-quality proteins, and their marine omega-3 fatty acids content [12,13]. Farmed salmon are also considered a sustainable alternative to terrestrial meat producing species, including beef cattle, hogs, and broiler chickens [14,15]. Salmonids are efficient livestock with low energy expenditure due to their cold-blooded homeostasis physiology and the gravitational constraints opposed by buoyancy under water [16,17]. Yet, despite these favorable characteristics, salmonid aquaculture faces a variety of environmental challenges. While aquaculture production addresses some of the issues of current food systems by using efficient marine livestock and (partly) moving production at sea, it does not address the underlying challenges of intensive and linear systems (inherited from agricultural practices). Future production increases will hinge on the industry's ability to solve challenges posed by parasites, diseases, nutrient emissions, and critical feed resource scarcity [18–20], among others. For instance, in Norway, the aquaculture industry has ambitions to increase salmonid production fivefold by 2050; however, it is increasingly clear that such a target cannot be achieved without identifying and addressing the underlying environmental issues of existing open sea-based systems [21].

Life cycle assessment (LCA) is one of the environmental accounting tools that can be used to provide the critical information necessary to improve the sustainability of such aquaculture systems. During the last 20 years, the emergence of LCA as a standardized method to measure the environmental impacts of products and services enabled researchers to assess complex seafood systems and provide sustainability guidelines to the industry and policymakers [22–24]. Since salmonids are one of the most high-value types of aquaculture species produced worldwide [1], and the interest to increase production volumes is high, this finfish type is the most studied species group in the aquaculture LCA literature. In fact, it is the only group for which LCA studies exist for various species, countries, feed recipes, and production technologies. LCA has been employed to compare products with similar functions [25], measure differences between feed formulations [26,27], and to evaluate the environmental efficiency of trout [28–30], and salmon [31–33]. Within this species group, the LCA methodology has also been employed repeatedly to compare the environmental impact of existing and emerging production technologies [34–36].

Following the accumulation of aquaculture LCA studies, several reviews recently analyzed this section of the literature, focusing on the application of the LCA methodology [37–39] and comparing production systems [40–43]. In the latter ones, attempts to compare results focused primarily on the comparison of the system's intensities (intensive, semi-intensive, and extensive) and culture types (polyculture vs. monoculture). Overall, the authors strictly avoided cross-study single factor statistical comparisons, because LCA data are known to be subject to a wide range of intricate influential variables. The scoping of this review has a sensitive balance, since a broad scope increases sample sizes and the influence of confounding factors, while a narrow one reduces discrepancies and the number of data points available. While previous reviews in this field focused on the whole aquaculture sector, we believe farmed salmonids to be a suitable species group to do an in-depth, comprehensive literature review. Such a scope allows us to study how different production systems for similar species with similar requirements compare, and if such LCA comparisons are meaningful. We consequently analyzed three key features of current salmonid LCAs. Firstly, we provide an overview and compare methodological choices. The objective is to evaluate common methodological practices, identify shortcomings, and suggest recommendations to practitioners. Secondly, we map and compare the inventory and modeling protocol of each study. We strive to identify the strengths and weaknesses of models and datasets used in publications. Thirdly, we perform a cross-study analysis on the life cycle impact assessment (LCIA) and FCR results through the prism of production technologies. Within this third analysis, we are making a first attempt to perform a cross-study statistical comparison of LCIA and FCR scores. This statistical protocol voluntarily ignores confounding factors to specifically investigate if LCA results comparisons can be performed on relatively homogenous samples (similar salmonid species) despite the influence of other variables. With this, our objectives are to assess if overarching statistical differences between groups can be identified, if discrepancies between studies are simply too great to validate such comparisons, and if LCA methodological developments are required to enable more direct comparability.

2. Materials and Methods

2.1. Study Selection

A systematic analysis of 24 LCA studies (20 peer-reviewed articles and four gray literature research) based on salmonid aquaculture systems was performed in this review (Table 1). The systems assessed produced Atlantic salmon (*Salmo salar*), rainbow trout (*Oncorhynchus mykiss*), Arctic char (*Salvelinus alpinus*), Chinook salmon (*Oncorhynchus tshawytscha*), brook trout (*Salvelinus fontinalis*), and brown trout (*Salmo trutta fario*). All of these species are classified under the Salmonidae family. Article identification was performed using the online search engines of Science Direct, Web of Science, Google Scholar, and Google. The keywords used were a combination of "LCA", "life cycle assessment", "aquaculture", "farmed salmon", "trout", "rainbow trout", and "marine products." The timespan ranged from 2000 to 2018. We identified 355 studies of interest. Primary refinement focused on titles and abstracts, with "environmental assessment" and "aquaculture system" as the selection criteria to isolate 65 publications. Secondary refinement concentrated on methods and models to select research focusing on salmonid aquaculture systems and containing original LCA case studies, resulting in the final selection of 24 assessments (Section 2.1.1, "article selection", supplementary data). A study was deemed original when significant first-hand data collection of at least one of the main foreground processes (hatchery and grow-out) took place. Five studies qualifying for selection were excluded. The inventories of Buchspies et al. [44], Ytrestøyl et al. [45], and Hall et al. [46] revealed an absence of original data. Winther et al. [47] was excluded in favor of Ziegler et al. [48] since both are presenting the same data/results and the latter is a peer-reviewed publication. Samuel-Fitwi et al. [49] was removed from the list in favor of Samuel-Fitwi et al. [50] since both studies were built with the same dataset and the latter assessed different production technologies (Section 2.1.2, "study exclusion", supplementary data).

2.2. Analytical Procedure

This work was articulated around three main analytical sections: (1) The specific choices related to LCA methodology, (2) the construction of life cycle inventories, and (3) a cross-study technological comparison based on compiled LCIA and FCR results (Section 2.2.1, "scope of the review", supplementary data). All analyses were performed in Excel. The first section presented the main methodological choices made by LCA practitioners. This encompassed mapping and comparing selections of functional units (FU), system boundaries, characterization methods, characterization factors, multi-functionality, as well as checking for the inclusion of interpretation tools such as contribution, sensitivity, and uncertainty analyses.

Table 1. List of articles and reports selected for review.

ID	Author and Year	Topic	Organization
1	Aubin et al. [51]	Carnivorous finfish production systems	INRA [a]
2	Avadi et al. [30]	Artisanal vs. commercial Peruvian feed	INRA [a]
3	Ayer et al. [52]	Comparison of copper and nylon net-pens	EarthShift Global [b]
4	Ayer and Tyedmers [34]	Culture systems in Canada	Jacques Witford [c]
5	Boissy et al. [29]	Impacts of plant-based salmonid diets	Montpellier U
6	Chen et al. [28]	Trout farming in France	INRA [a]
7	D'Orbcastel et al. [53]	Comparison of two trout systems	IFREMER [d]
8	Dekamin et al. [54]	Rainbow trout production in Iran	UMA [e]
9	Ellingsen and Aanon. [25]	Comparison of wild cod, farmed salmon, and chicken	SINTEF [f]
10	Grönroos et al. [36]	Finnish cultivated rainbow trout	SYKE [g]
11	Hognes et al. [32] *	Norwegian salmon production	SINTEF [f]
12	Liu et al. [35]	Carbon footprint of two farming models for salmon	SINTEF [f]
13	McGrath et al. [33]	Novel close aquaculture salmon technology	Dalhousie U
14	Newton and Little [55]	Farmed Scottish salmon	Stirling U
15	Nyhus [56] *	Comparing salmon closed and open cage system	NTNU [h]
16	Papatryphon et al. [24] *	Trout farming in France	INRA [a]
17	Parker [57]	Implications of high animal by-product feed inputs	British Colum. U
18	Pelletier et al. [31]	Global salmon farming systems	Dalhousie U
19	Samuel-Fitwi et al. [50]	Raising rainbow trout in different systems	GMA [i]
20	Silvenius et al. [58]	Climate and eutrophication impact of Finnish trout	LUKE [j]
21	Smárason et al. [27]	Icelandic arctic char fed three different feeds	Matis [k]
22	White [59] *	Efficiency of the Tasmanian salmon industry	Bond U
23	Wilfart et al. [60]	Accounting of aquaculture systems	INRA [a]
24	Ziegler et al. [48]	Carbon footprint of Norwegian seafood products	SIK [l]

[a] INRA = National Institute for Agricultural Research; [b] EarthShift Global = sustainability consultancy company; [c] Jacques Witford = environmental consulting engineering company; [d] IFREMER = French Research Institute for the Exploitation of the Sea; [e] UMA = University of Mohaghegh Ardabili; [f] SINTEF = The Company for Industrial and Technical Research; [g] SYKE = Finnish Environmental Institute; [h] NTNU = Norwegian University of Science and Technology; [i] GMA = Society for Marine Aquaculture; [j] LUKE = Natural Resource Institute Finland; [k] Matis = Icelandic food and biotechnology R&D institute; [l] SIK = Swedish Institute for Food and Biotechnology; and U = University. * Non-peer reviewed reports.

The second section measured and compared the data quality of inventories by reviewing the main foreground and background sources used by authors. We also estimated study completeness by mapping process inclusion and exclusion, including an evaluation of the LCA practitioner reporting practices. In this analysis, the literature was divided into two groups: Trout and salmon systems. This separation was performed to highlight significant inventory differences between the trout and salmon LCAs, and its application was strictly limited to this section. Overall, 14 independent processes were identified to fully cover the life cycle of salmonids (Table 2).

In the third section, LCIA and key inventory data in terms of FCR results were compared across studies according to their production technology. Although FCR scores are LCI data and as such differ significantly in nature from LCIA results, we found it meaningful to add this parameter to the analysis. The FCR indicates the quantity of feed use per unit of fish produced, and although it has been criticized lately in cross-species efficiency comparison [61], we estimated this parameter to be representative of the overall efficiency of salmonid production systems. It is also a parameter directly linked to the aquafeed process, which is by far the largest contributing component to the life cycle emissions of salmonids [51,54,57]. FCR has recently been identified as a useful indicator to follow up environmental performance in farmed aquaculture within a species and using the same feed [62]. While it was not possible to standardize feed compositions across studies due to a lack of inventory disclosure, our comparison dealt with species of the same family, with similar nutrient requirements to enhance comparability of FCRs. For the LCIA data, the comparison was performed for the four most commonly used impact categories in the reviewed studies: Global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), and cumulative energy demand (CED). Out of the 24 LCA studies selected, a total of 67 scenarios were identified. These LCA scenarios have variable focus, among others investigating impacts of different feed composition, electricity mix, or cage structure. Scenarios were selected based on their production technologies. Scenarios outside the scope of this review were excluded. When scenarios used similar production technology and only the cofounding factor differed, results were averaged. After selection, 35 scenarios remained for GWP, 30 for AP, 29 for EP, and 24 for CED. Of those scenarios, 33 disclosed their FCR scores, which were subsequently used in this analysis (Section 2.2.2, "LCIA scenario selection", supplementary data).

Table 2. List of salmon and trout life cycle processes.

Process	Process Description
Energy carrier	All energy carriers required through the life cycle (e.g., electricity, gas)
Transport	All transports required through the life cycle, without distribution (e.g., lorry, shipping)
Chemotherapeutant	Includes chemicals, disinfectants, and veterinary products
Equipment	Supply considered having a lifetime ranging from 1 to 10 years
Infrastructure	Supply considered having a lifetime greater than 10 years
Feed production	Agricultural and fishery ingredients, and their transformation into aquafeed
Egg production	Selection, and reproduction of broodstock for egg production
Hatchery	Grow-out of eggs into 70 g large juvenile fish
Fish production	Grow-out of juveniles into 4–5 kg adult fish
Effluent	Emissions of feces, urine, and feed waste generated during grow-out
Effluent treatment	Treatment of hatchery and fish production effluents
Infrastructure EOL	Dismantling and waste treatment of farms infrastructure
Processing	Processing of live fish into HOG or filleted fishes
Distribution	Transport and retail of HOG or filleted fishes from processing to customers

While the comparability of LCIA results is uncertain due to the wide range of diverging parameters existing between studies (e.g., year, country, production site, feed composition, characterization method, assumption, etc.), we increased consistency by only comparing cradle-to-gate systems producing 1 t live-weight salmonids. Most studies already expressed their results in this format. Six studies used different system boundary and FU combinations [35,36,48,52,55,58]. We performed a small system boundary adjustment of Ayer et al. [52] by adding hatchery impacts using contribution analysis results from Pelletier et al. [31]. Cradle-to-farm gate impacts of 1 t FU produced were available in Liu et al. [35] and Ziegler et al. [48] and part of the secondary results. Newton and Little [55], Silvenius et al. [58], and Grönroos et al. [36] adjusted impacts were obtained through personal communication with the authors. Grönroos et al. [36] adjusted results differ significantly from the published results due to an inventory error discovered by the authors (Section 2.2.3, "adjustment calculations", supplementary data).

All scenarios were separated into four production technology clusters. Salmonid aquaculture production technologies revolve around two major characteristics: Sea-based vs. land-based (clusters A-B) and open vs. closed (clusters C-D) systems. This led us to the selection of four-production technology clusters: (1) Closed sea-based systems, (2) open land-based systems, (3) open sea-based systems, and (4) closed land-based systems. Cluster (1) contains experimental pilots of either a solid wall aquaculture system [33,56] or marine floating bag [34,36]. These systems provide a controlled production environment at sea and a form of waste collection. Cluster (2) exclusively gathers flow-through aquaculture system (FTAS) technology. Such raceways usually divert flowing water (often from rivers using gravity) through successive concrete flumes before discharging back to the water body [63]. While this is mostly a freshwater system used for rainbow trout production, a few studies also investigated the production of salmon and char using marine and brackish waters [27,34]. Cluster (3) is the most common form of salmon production, usually taking place in net pens of various sizes and forms (mostly circular). In open cages, only nets separate the biomass from the environment, which means that salmonid metabolites and extra feed pellets are directly emitted to the surrounding waters [64]. Cluster (4) is mostly recirculating aquaculture system (RAS) technology. This land-based design uses circular tanks to produce trout, char, and salmon, primarily in freshwater conditions. Water movement and oxygenation is insured by machinery and is recycled using drum and bio-filters [65].

2.3. Statistical Analysis

A statistical analysis of each production clusters for LCIA and FCR scores was performed in SPSS v24 (IBM Corp, Armonk, New York, United States). In this review, we deliberately opted for a single parametric statistical protocol to compare the means of the different technological clusters. This is a first attempt to make a statistical LCA cross-study comparison without involving multifactorial analysis to investigate if a simpler, non-discriminant, comparative protocol can be applied to identify overarching trends in LCA results. We are not trying to compare the groups independent of confounding factors, but to investigate if current LCA data can be statistically compared despite them. The equivalent non-parametric test to ANOVA (Kruskal-Wallis) was disqualified due to heterogeneous distributions in the data. The LCIA and FCR data was analyzed following a four-group comparison (clusters 1 to 4), and two subsequent two-group comparison testing data of clusters A-B and C-D (Figure 1). Throughout the analysis, the significance level α (referring to the probability of making a type one error) was set to 5%. An identical test protocol was applied to all data. Each group was tested for normality with Shapiro-Wilk and homogeneity of variances using a Levene's test. For Shapiro-Wilk, p-values were tested against the null hypothesis, assuming a normal distribution within clusters, while for Levene's test, the null hypothesis was supporting similar group variances. If homogeneity of variance was achieved, a one-way ANOVA t-test was used to compare the means of clusters. Where variances were found to be statistically different, a Welch t-test was performed to compare the homogeneity of the means. If the Welch-test indicated a statistically significant difference between some of the means, the Games-Howell post hoc test was performed to identify which of the cluster's means were different from one another. Results of both the ANOVA and Welch t-test were rejected if the group involved displayed were skewed. For these three statistical tests, the null hypotheses assume means equality.

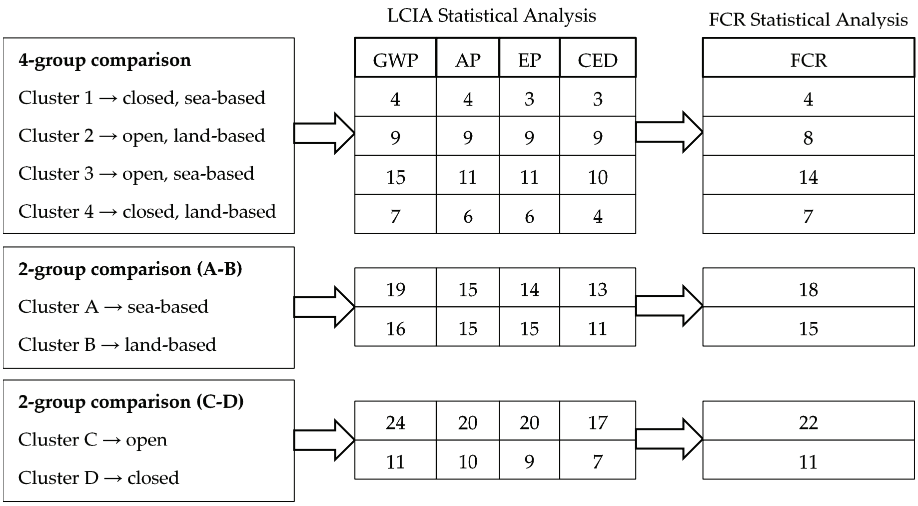

Figure 1. Structure of the life cycle impact assessment (LCIA) and feed conversion ratio (FCR) statistical analysis displaying the numbers of data points per clusters.

3. Results

3.1. Methodology Analysis

The location of the production systems reflects the current dominance of salmonid aquaculture by a handful of industrialized countries; Norway, the United Kingdom, Canada, Chile, and Australia dominate salmon production, while Finland and France concentrate on trout species. Dealing with multi-functionality remains controversial and heterogeneous, with practitioners using a mix of mass, economic, and energy-based allocation throughout the literature (Table 3). System expansion, recommended by ISO 14040 [66], is rarely employed. All studies follow the attributional LCA framework, except for Samuel-Fitwi et al. [50], employing the consequential approach. Out of the 24 studies reviewed, 20 are using different versions of SimaPro (PRé Sustainability, Amersfoort, the Netherlands) for modeling and calculations. CML-IA remains the most common characterization method (applied 14 times), followed by the ReCiPe midpoint framework (applied five times). All studies provide a process-based contribution analysis, and most of them include a sensitivity analysis to test for critical data sources and/or methodological choices. Only four studies [30,33,52,54] included an uncertainty analysis.

The range of midpoint impact categories selected is widespread across studies. GWP, EP, AP, and CED are by far the most represented categories with application rates of respectively 100%, 87%, 78%, and 61%. Characterization factors used to calculate these midpoint scores evolve over time, but are mostly (in frequency) based on the work of IPCC [67], Heijungs et al. [68], Huijbregts [69], Frischknecht [70], and Frischknecht et al. [71]. LCA practitioners often add biotic resource use (43%), water dependence (43%), and land occupation (39%) to measure additional impacts significant for salmonid aquaculture production. Results show strong discrepancies between the characterization methods, the characterization factors and their application by authors. For example, in CML-IA, the EP characterization factors (based on Heijungs et al. [68]) refer to phosphorus (kg PO4-e) without discerning marine and freshwater nutrient limitation, while ReCiPe expresses the same impacts based on nitrogen (kg N-e) and phosphorus (kg P-e) depending on the emissions medium [72]. In different studies, water dependence is also named water depletion [30], water use [29,59], freshwater footprint [32], and consumptive water use [55]. Land occupation is inconsistent in both naming and units between studies.

Table 3. Main methodological choices used in salmonids life cycle assessment (LCA).

ID	Goal and Scope Definition				LCIA/Interpretation			
	Functional Unit	Location	System Boundaries	Multi-functionality	CM	CA	SA	UA
1	1 t LW Rainbow Trout	France	Farm to farm	NA	CML-IA	✔	✔	
2	1 t LW Rainbow Trout	Peru	Cradle to farm	Energy allocation	CML-IA/ReCiPe	✔	✔	✔
3	1 t LW Atlantic Salmon	Chile	Farm to farm	Energy allocation	ReCiPe midpoint	✔	✔	✔
4	1 t LW Atlantic Salmon/Arctic Char	Canada	Cradle to farm	Energy allocation/SE	CML-IA	✔	✔	
5	1 t LW Atlantic Salmon/Rainbow Trout	Scotland/France	Cradle to farm	Economic allocation/MA	CML-IA	✔	✔	✔
6	1 t LW Rainbow Trout	France	Cradle to farm	Economic allocation	CML-IA	✔	✔	
7	1 t LW Trout/Arctic Char	France	Cradle to farm	NA	CML-IA	✔	✔	
8	1 t LW Rainbow Trout	Iran	Cradle to farm	NA	CML-IA	✔	✔	✔
9	200g Fillet Atlantic Salmon	Norway	Cradle to distribution	NA	Eco-indicator 99	✔	✔	
10	1 t Un-gutted Rainbow Trout	Finland	Cradle to processing	NA	Finnish factors	✔		
11	1 kg LW Atlantic Salmon	Norway	Cradle to farm	Mass allocation	ReCiPe midpoint	✔		
12	1 kg HOG Atlantic Salmon	Norway/USA	Cradle to distribution	Mass allocation	NA	✔		
13	1 t LW Chinook Salmon	Canada	Cradle to farm	Energy allocation	ReCiPe midpoint	✔	✔	✔
14	1 t HOG Atlantic Salmon	Scotland	Farm to processing	Economic allocation/MA	CML-IA	✔	✔	
15	1 t LW Atlantic Salmon	Norway	Cradle to farm	NA	ReCiPe midpoint	✔	✔	
16	1 t LW Rainbow Trout	France	Cradle to farm	Economic allocation	CML-IA	✔		
17	1 t LW Atlantic Salmon	Australia	Cradle to farm	Energy allocation/SE	CML-IA	✔	✔	
18	1 t LW Atlantic Salmon	NO/UK/Canada/Chile	Cradle to farm	Energy allocation	CML-IA	✔	✔	
19	1 t LW Rainbow Trout	Germany/Denmark	Cradle to farm	System expansion	CML-IA	✔	✔	
20	1 t Fillet Rainbow Trout	Finland	Cradle to distribution	Economic allocation/MA	Individual factors	✔		
21	1 kg LW Arctic Char	Iceland	Cradle to farm	Mass allocation	CML-IA	✔	✔	
22	1 t HOG Atlantic Salmon	Australia	Cradle to processing	Mass allocation	CML-IA	✔	✔	
23	1 t LW Atlantic Salmon	France	Cradle to farm	Economic allocation	CML-IA	✔	✔	
24	1 kg HOG Atlantic Salmon	Norway	Cradle to distribution	Mass allocation	Individual factor	✔	✔	

Abbreviations: CA = Contribution Analysis; CM = Characterization Method; HOG = Head-On-Gutted; LW = Live-Weight; MA = Mass Allocation; NA = Not Available; NO = Norway; SA = Sensitivity Analysis; SE = System Expansion; UA = Uncertainty Analysis; and UK = United Kingdom.

It is alternatively referred to as agricultural land occupation [30], surface use [53], land use [54,55], and land competition [50,60], and is expressed in square meters per year [28–30,50,54,55,60] or square-meters [53] alternatively. Finally, biotic resource use is also called net primary production in about one-fifth of the studies [28,29,51,53,59] (see supplementary data, Section 3.1.1).

3.2. Inventory Analysis

3.2.1. Trout Production Systems

First-hand aquaculture data originate primarily from French (Vivier de France SA, Murgat SAS) [29,51,53] and Finnish (Nordic Trout, Rehuraisio Ltd.) [24,58] trout farms. Le Gouessant cooperative [24,51], Marine Harvest [29], Biomar [58], and Cargill [50] are also frequent sources used for aquafeed modeling. Second-hand data covers a broader range of processes and is composed of a multitude of sources. Yet, cross-referencing and recurrent datasets can still be identified. Several practitioners use the data from Agreste [24,28,29] (French agriculture database) as well as Pelletier et al. [31], and Boissy et al. [29] to complete their feed and aquaculture dataset [28,30,54]. Various versions of the Ecoinvent database (https://www.ecoinvent.org/) are also used to model agricultural feed ingredients [29,58], as well as transports [28,30,50] and energy carriers [28,30,50]. Thrane [73], Schau and Fet [74], and Vázquez-Rowe et al. [75] are occasionally used in Boissy et al. [29] and Fitwi et al. [50] to model impacts from wild fisheries associated with the production of fish meal.

A vast majority of trout studies exclude egg production (broodstock reproduction), effluent treatment, processing, and distribution. Out of 10 studies reviewed, the inclusion frequency of these processes was 30%, 40%, 20%, and 10% (Figure 2). Reasons for these exclusions vary from the absence of reliable technology and legislation (e.g., effluent treatment) to the selection of narrower system boundaries (e.g., processing, distribution), and the estimated negligible impacts of a given process (e.g., egg production). None of the trout LCAs included the infrastructure end-of-life (EOL) process because of its versatility and estimated overall low environmental contribution (cut-off criterion). The inclusion of equipment and infrastructure and chemotherapeutants are relatively high, with a use frequency of 70%. Equipment and infrastructure construction of trout systems is intensive since this species is mostly farmed with FTAS and RAS land-based technology. In the literature, the use of chemicals and veterinary products is often reported under the overarching term "chemotherapeutant". However, in practice, this process has an uncertain connotation. In some studies, it refers to chemicals, disinfectants, limestone, and oxygen used in the hatchery and grow-out processes, while in others it also includes veterinary treatments such as antibiotics. Overall, energy carrier, transport, feed production, hatchery, fish production, and effluent are the most common processes (Figure 2). Practitioners expect these processes to have the highest contribution, which makes them less prone to cut-offs, system boundaries, and FU exclusions (see supplementary data, Sections 3.2.1.1 and 3.2.1.2).

3.2.2. Salmon Production Systems

Like for trout, first-hand salmon LCA data collections concentrate on fish and feed production and (to a lesser extent) on the hatchery process. In several cases, data from the grow-out phase is confidential and prevent the disclosure of the company's name [25,31,32,34,35,48,55,57]. Few studies openly report using grow-out data from Agrimarine Inc [33], Marine Harvest [56], and the companies Tassal, Huon Aquaculture, Petuna, Van Diemen aquaculture, and Saltas [59]. Sources of aquafeed manufacturers are more frequent, with large corporations like Biomar [32,56], Skretting [32,56], Cargill [32,33,56], and smaller firms like Taplow feeds [33], and Ridley [59] being cited more often by name. The spectrum of the second-hand data used in salmon LCA is similar to that for trout. The Ecoinvent database is often used to complete transports [27,31–35,48,52,55–57], energy carriers [27,31,32,34,35,52,55–57], and agricultural feed ingredient [27,32,33,48,56,59] models. In more recent years, the Agri-footprint database (http://www.agri-footprint.com/) has also been used similarly by practitioners, mostly to improve the quality of their aquafeed datasets [32,35,57]. Data published by Pelletier et al. [31] is cross-referenced throughout the reviewed literature and appear to be central for several aquafeed models [31,33,34,52,56,59]. Like for trout, Thrane [76], Schau and Fet [74], and Tyedmers [77] are sporadically used in Ellingsen and Aanondsen [25], Pelletier et al. [31], and Ziegler et al. [48] to model capture fisheries. In recent work, Parker [57] used Cashion et al. [78] and Parker and Tyedmers [79] to cover similar processes.

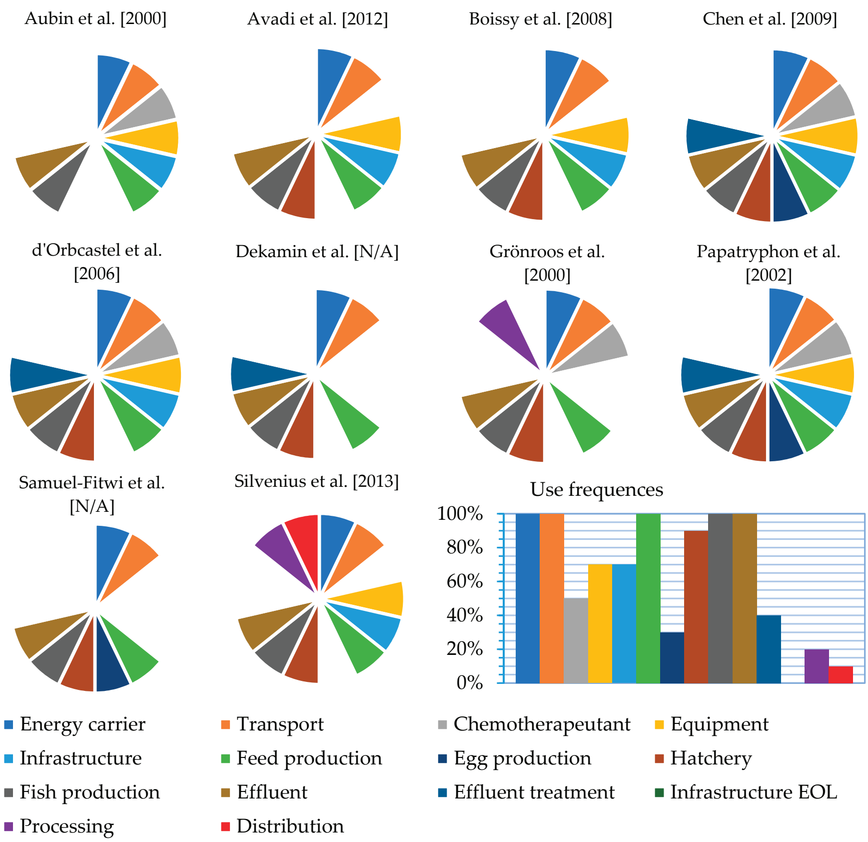

Figure 2. Process inclusion and exclusion in trout LCA studies (data collection dates are placed in brackets).

In the life cycle inventories of salmon LCA models, energy carrier, transport, feed production, hatchery, fish production, and effluent processes have a very high inclusion rate (Figure 3). Only hatchery and effluent are under the 100% inclusion with 86% and 79%, respectively. These lower rates can be explained by few process exclusions due to narrower system boundaries (e.g., hatchery) and/or to impact categories selection (e.g., effluent). As such, only carbon footprint studies [32,35,48] excluded effluent since it is estimated not to affect GWP. Chemotherapeutant, equipment, infrastructure, egg production, effluent treatment, infrastructure EOL, processing, and distribution have lower inclusion rates ranging from 7% to 50%. Here again, processing and distribution directly depend on the study's scope and its system boundaries. Infrastructure EOL is included in Ayer et al. [52] to specifically compare two types of infrastructure through their life cycle (nylon and copper net-pens). Effluent treatments are systematically excluded from open net-pen cages studies but are described by Ayer and Tyedmers [34] and McGrath et al. [33] in the modeling of closed sea-based systems equipped with filters (see supplementary data, Sections 3.2.2.1 and 3.2.2.2).

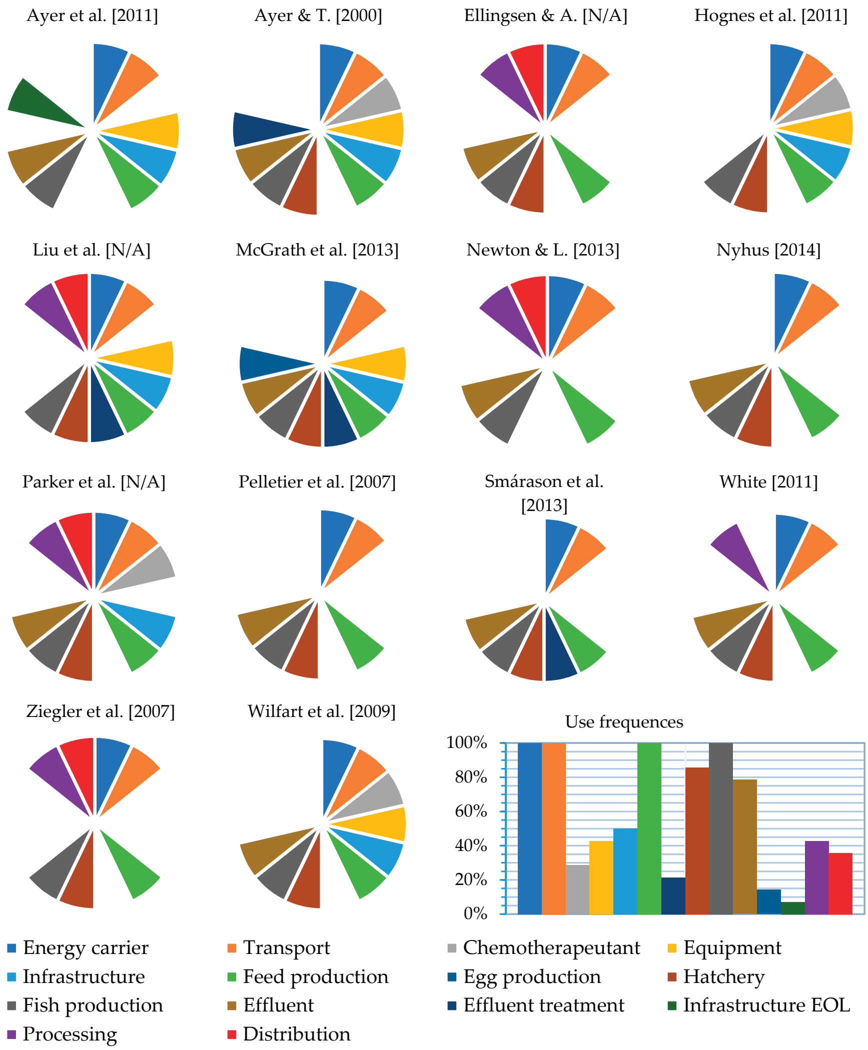

Figure 3. Process inclusion and exclusion in salmon LCA studies.

3.3. *Cross-Study Technological Comparison*

3.3.1. Overview of LCIA and FCR Results

GWP impacts vary widely across the reviewed literature with a delta between the maximum (13,622 kg CO_2-e) [53] and the minimum (1157 kg CO_2-e) [54] reaching 12,465 kg CO_2-e (Figure 4). Within clusters 1 to 4, the situation is also heterogeneous with delta scores of respectively 1585; 4252; 7170; and 11,799 kg CO_2-e. Overall, closed sea-based systems perform best with an average score of 2404 kg CO_2-e per ton LW salmonids produced at the farm gate. Open land-based systems rank second with an average impact of 2613 kg CO_2-e while open sea-based and closed land-based systems arrive in 3rd and 4th position with 2933 and as much as 6414 kg CO_2-e for the same production output. FCR values seem to fluctuate independently of climate change impacts (Figure 4).

AP impacts are also heterogeneous. Grönroos et al. [36] reported as little as 8.8 kg SO_2-e per ton LW trout while Ayer and Tyedmers [34] calculated a score as high as 63.4 kg SO_2-e for a similar output (Figure 5). Intra-cluster heterogeneity is comparable to that of climate change impacts. Deltas of 10.1 kg SO_2-e, 23.3 kg SO_2-e, 33.6 kg SO_2-e, and 51.5 kg SO_2-e are observed between minimums and maximums of clusters 1, 2, 3, and 4 respectively. Closed land-based systems perform worst with 26.7 kg SO_2-e, while closed sea-based systems outperform all other clusters by generating 15.1 kg SO_2-e per FU on average. Open land-based and sea-based systems reach the 2nd and 3rd positions with average scores of 16.3 kg SO_2-e and 18.7 kg SO_2-e. Graphic representation of the FCR seems to fluctuate independently of AP impacts scores (Figure 5). For instance, Samuel-Fitwi et al. [50] calculated emissions of 40.7 kg SO2-e for an FCR of 0.86 while Dekamin et al. [54] reported an FCR of 1.47 with emissions as low as 18.7 kg SO2-e.

Out of 29 EP scores extracted from LCA scenarios, values ranged from 4 kg [50] to 84 [57] kg of PO_4-e, generating a difference of factor 21. Results suggested that closed systems perform best, emitting only 17.3 kg PO_4-e (land-based) and 26.7 (sea-based) kg PO_4-e. Open systems generated on average 47.3 kg PO_4-e (sea-based) and 50.6 (land-based) kg PO_4-e into fresh and marine water bodies (Figure 6). Intra-cluster deltas remain high with differences ranging from 30.3 kg PO_4-e, 51.2 kg PO_4-e, and 62.6 kg PO_4-e for systems 4, 2, and 3 respectively. Closed sea-based systems are the exception, displaying a low delta of 8.6 PO_4-e for this impact category. Any direct correlation between FCR and EP impacts remain widely uncertain with studies displaying both low impacts and high FCR [34,54] as well as low FCR and high impacts [55,57].

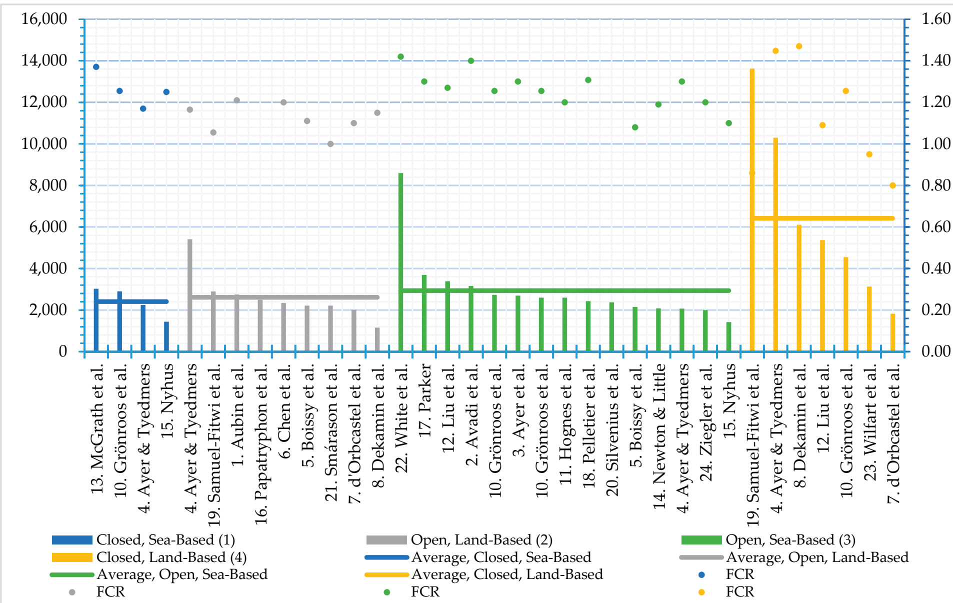

Figure 4. Salmonids global warming potential (GWP) impacts (kg CO2-e) and FCR based on production technology clusters.

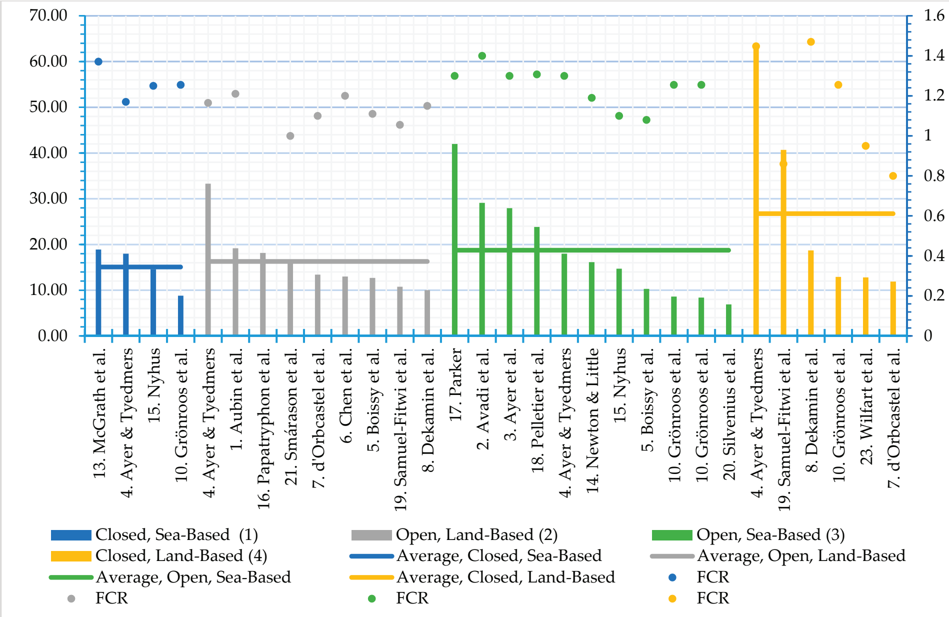

Figure 5. Salmonids acidification potential (AP) impacts (kg SO2-e) and FCR based on production technology clusters.

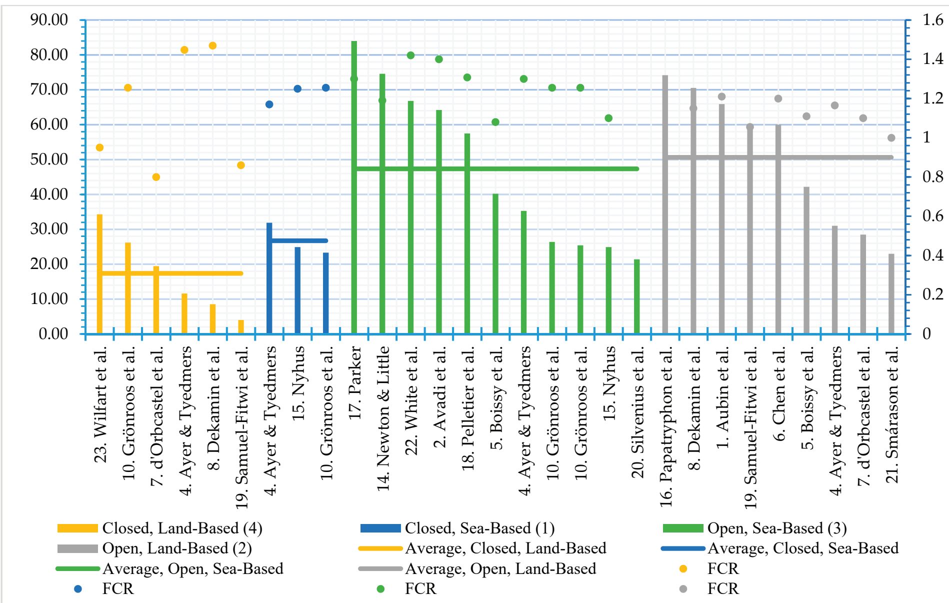

Figure 6. Salmonids eutrophication potential (EP) impacts (kg PO4-e) and FCR based on production technology clusters.

Variability of CED scores is substantial throughout the reviewed literature. The delta for this category represents 206,100 MJ-e, which corresponds to a difference of factor 9.23 between the maximum of 233,000 MJ-e [34] and the minimum of 26,900 MJ-e [34] (Figure 7). Intra-cluster impact deltas range from 172,569 to 29,764 MJ-e with closed, land-based systems reporting the most variable scores (indicating improvement potential). Closed sea-based systems displayed the most consistent results. Salmonid life cycle CED results show that land-based systems have the highest energy requirements with cluster averages equal to 133,220 MJ-e (closed) and 75,943 MJ-e (open). Sea-based systems, on the other hand, perform best with average CEDs of 37,913 MJ-e (open) and 54,620 MJ-e (closed) per ton LW salmonids produced at the farm gate (Figure 7). Again, a high FCR does not appear to correlate with high CED directly. Studies reporting low FCR and high CED [27,60] and some displaying the inverse relationship [30,59] are commonly found.

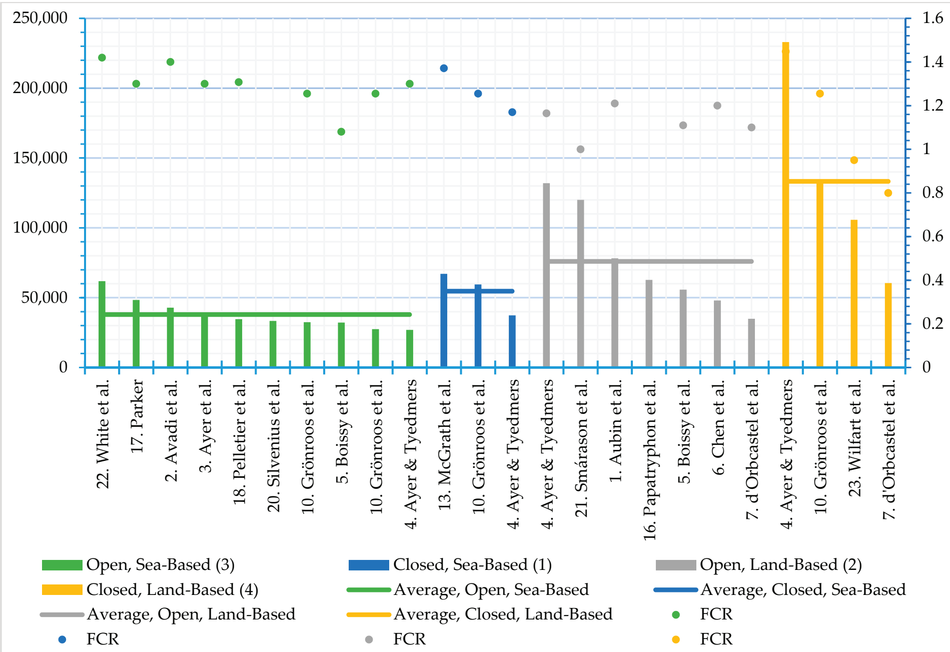

Figure 7. Salmonids cumulative energy demand (CED) impacts (MJ-e) and FCR based on production technology clusters.

The average FCR values of clusters 1, 2, 3, and 4 are 1.262, 1.124, 1.256, and 1.125, respectively for GWP. While FCR fluctuates slightly across impact categories due to differences in sample sizes, results consistently show that open and closed land-based clusters have the lowest average FCR with scores ranging between 1.124–1.131 and 1.113–1.131 and that closed sea-based and land-based technologies oscillate in the upper range within intervals of 1.225–1.265 and 1.249–1.291 (Table 4). Since FCR averages variations between impact categories are minimal and only imputable to sample size, the sub sequential statistical four-groups and two-groups analysis of FCR averages were performed independently of impact categories variations, on the single largest sample size available (also corresponding to the FCR GWP group).

Table 4. Overview of clusters average FCR for each impact categories.

Cluster	Technology	FCR (GWP)	FCR (AP)	FCR (EP)	FCR (CED)
1	Closed, Sea-Based	1.262	1.262	1.225	1.265
2	Open, Land-Based	1.124	1.124	1.124	1.131
3	Open, Sea-Based	1.256	1.249	1.261	1.291
4	Closed, Land-Based	1.125	1.131	1.131	1.113

3.3.2. Statistical Comparison of LCIA and FCR Results

(a) Four-Groups Analyses

Out of the 16 LCIA groups (Figure 1), cluster 2-3 for GWP and 2-4 for AP failed normality tests. The remaining clusters displayed normal distributions. Levene's results showed that all groups violated the null hypothesis, pointing to heterogeneous variances between groups. GWP and AP Welch t-test means comparison were rejected because of the skewed distributions of some of their clusters. This means that 50% of the data is not comparable according to this statistical protocol due to non-normal distributions. The means comparisons were validated for EP and CED. The test revealed statistically significant differences of the means between the clusters of EP (p = 0.004) but no differences were found between the clusters' means of CED (p = 0.066). Post hoc Games-Howell test identified the statistical difference within EP to be between clusters 1-2, 3-4, and 2-4. It means that overall, only significant differences were identified for EP between closed, sea-based and open, land-based systems (p = 0.030), between open, sea-based and closed, land-based systems (p = 0.012), as well as open, land-based and closed, land-based systems (p = 0.006). For the four FCR groups (Figure 1), each cluster validated the condition of normality and results showed clear heterogeneity of variances (p = 0.0001). The null-hypothesis of Welch was rejected (p = 0.023) indicating a significant difference between at least two of the clusters' means. Post hoc testing indicated that the difference was lying between cluster 2-3 (p = 0.009; open, land-based and open, sea-based systems) while the remaining of the clusters had statistically equal means (see supplementary data, Sections 3.3.2.1 and 3.3.2.2).

(b) Two-Groups Analysis

Two two-group analyses were performed on both LCIA and FCR data. One was testing the data of sea-based (cluster A) vs. land-based (cluster B) studies while the other was testing open (cluster C) vs. closed (cluster D) systems. Each two-group LCIA analysis consists of eight clusters while each FCR one consists of two (Figure 1). Cluster A-B for GWP, B for AP, and A for EP failed the normality test. Homogeneity of variances was observed in AP and EP while the null hypothesis was rejected for GWP and CED. GWP, AP, and EP ANOVA and Welch means comparisons were rejected due to skewness in at least one of their clusters. This means that 75% of the data could not be tested and interpreted. The Welch t-test mean comparison for CED was accepted, showing significant differences between the means of closed, sea-based and open, land-based clusters for this impact category (p = 0.01). All clusters failed normality, except cluster D for EP in the open vs. closed data comparison. Levene's test demonstrated that clusters of GWP, EP, and CED had heterogenous variances while AP clusters displayed a homogeneous one. Since seven out of the eight clusters were skewed, results of both ANOVA and Welch t-tests were rejected. In this case, 100% of the data was rejected and could not be compared using this statistical protocol (see supplementary data, Sections 3.3.2.3 and 3.3.2.4). The FCR data of clusters A-B confirm the null hypothesis of Shapiro-Wilk, supporting that both samples are distributed normally. The same results could be observed for the FCR data of clusters C and D. Also, both A-B (sea-based and land-based) groups and C-D (open and closed) systems displayed heterogeneous variances. In the A-B comparison, the null hypothesis of Welch was rejected (p = 0.021), demonstrating a significant difference between the means of sea-based and land-based clusters. Opposite results were found for the C-D comparison (p = 0.654); open and closed systems FCR means were not statistically different from each other (Section 3.3.2.5, "FCR two-group stat. A-B", and Section 3.3.2.6, "FCR two-group stat. C-D", supplementary data).

4. Discussion

4.1. Methodologies

The methodological variations identified in this LCA review are understandable considering the difficulty of developing a "one size fits all" LCA methodology for production systems that differ in time, space, and structure. Overall, the results showed relatively consistent FU and system boundaries but displayed variations in the way LCA practitioners deal with multi-functionality and LCIA. These variations were not necessarily negative since methods selections were often case-specific, determined by the scope of the study, and could also be driven by methodological developments. Keeping this flexibility enables practitioners to adapt the method to the production system instead of the opposite. However, method customization should not be achieved at the expense of clarity. For instance, for impact category selection, it is essential to cover the main environmental impacts of the system while also limiting the number of categories to keep the results comprehensible for readers. This issue is discussed in Steinmann et al. [80], who found that within current impact categories, seven indicators were estimated to cover 92% of the variance of product rankings from the Ecoinvent database (based on 976 products). It means that by selecting GWP, land use, ozone depletion, AP, EP, marine, and terrestrial ecotoxicity, the practitioner covers nearly the entire spectrum of impacts at the midpoint level. The authors also suggest using RACER (relevant, accepted, credible, easy, robust) criteria to refine the indicator selection, as well as encourage the use of regionalized impact categories [80]. Based on Steinmann's perspective and the categories selection rates in salmonid LCA, we recommend practitioners assessing such aquaculture systems to concentrate primarily on: (1) GWP, (2) EP, (3) AP, (4) CED, (5) marine ecotoxicity, (6) terrestrial ecotoxicity, and (7) land-use. Biotic resource use and water use are also of interest, depending on the scope of the study. However, while these established impact categories are adapted to measure impacts of product and services, they are not necessarily well equipped for production technology comparison. The development of aquaculture specific impact categories is long wished for in the literature [40,50,54,58] and suggestions to include local ecological impacts have already been made years ago [81]. In this perspective, accounting for the impacts of aquaculture on biodiversity and water quality suggested by Ford et al. [81] would significantly improve the environmental criteria by which we compare the different types of salmonid aquaculture production systems.

There is an overall lack of innovative methodological integration in LCA of salmonids production systems (Table 3). For instance, it is unclear why several recent publications [27,54,57] continue to apply characterization factors from CML 2001 [68,82,83] when new (and more comprehensive) indicators are available in ReCiPe [72,84,85]. Lack of comprehensive and standardized methodological reporting is another shortcoming of the current literature. For example, since no rules exist for including and reporting contribution, sensitivity, and uncertainty analyses, the quantity and quality of information provided by LCA practitioners vary widely. While contribution analysis is commonly disclosed in LCA studies (although not always in numerical terms), the inclusion of uncertainty analysis remains rare. In this review, only one out of five LCIA results were calculated with an uncertainty range (Table 3). Systematic uncertainty assessments would benefit the comparisons conducted in this paper. The lack of overall methodological description is a challenge for replicability, transparency, and review. For instance, gathering the detailed characterization factors used for each impact category was complicated by the overall lack of data description (both in articles and supplementary data). We recommend that LCA practitioners pursue a flexible application of the LCA methodology to best suit each study's scope. It should entail considering the latest methodological LCA developments (when relevant) and going further than the LCA guidelines provided by the ISO framework [86] using, for instance, the procedure developed by the European Research Commission [87]. However, this adaptive application of LCA necessitates detailed and systematic reporting to be fully exploitable (reproducible and comparable) by the research community. It is therefore critical that researchers document their LCA methodological choices extensively, especially in the supplementary data. We recommend practitioners to systematically disclose and argue for major modeling decisions (e.g., characterization factors) and assumptions (e.g., cut-off criteria) performed under their assessment.

4.2. Inventories

In both trout and salmon LCA studies, process completeness is directly influenced by the FU and system boundaries selection (e.g., hatchery, processing, distribution), by the choice of impact categories (e.g., effluent), by the type of technology studied (e.g., effluent treatment), and by the scope of the research (e.g., infrastructure EOL). The inclusion of equipment and infrastructure is lower in salmon LCI, which could be explained by the "light-weight" of open sea-based cages compared to RAS and FTAS systems, more frequently used in trout production. The inclusion of chemotherapeutic agents appears more frequently for trout than salmon (55% and 23%, respectively). This could be due to the importance of water quality treatments (input of oxygen, limestone, and other chemicals) in FTAS and RAS production systems. Often the exclusion of processes such as chemotherapeutants, equipment, infrastructure, and egg production is based on the assumption that those processes have an overall low contribution [35,54,55]. Since process-based LCA remains resource intensive to conduct, practitioners often choose to perform such cut-offs, especially if data availability is low and/or quality poor. Interpreting the importance of such processes (when accounted for) is also challenging due to their variable definitions across studies. For instance, several authors include the production of salmonid eggs in their inventory [27,28,33,50], but the specific nature of this process remains uncertain because detailed descriptions are lacking. Considering the growing biological challenges faced by open sea-based salmon production [88,89], accounting for the environmental impacts of parasites and diseases treatments is becoming a necessity. So far only three of the 13 salmon LCAs reviewed accounted for a form of chemotherapeutant use (antibiotics and water quality treatments), and only for closed land-based production phases (hatcheries using FTAS or RAS systems) [32,34,57]. The frequent salmon lice sea-based treatments using biological (cleaner fish), chemical (baths and feed), and mechanical (primarily warm-water baths) processes remain unaccounted for in current LCAs. In Norway, use of cleaner fish requires aquaculture and fisheries activities dedicated to produce and capture Lumpfish (*Cyclopterus lumpus*) and various species of Wrasses (*Labridae*). The value-chains involved are complex and require significant use of marine ingredient rich aquafeed to produce the cleaner fish and sustain them throughout the salmon production cycle. Overall, the use of chemicals is following a descending trend, mainly because of sea-lice resistances. Yet, the use of hydrogen peroxide and other substances like azamethiphos and cypermethrin in baths remain commonly used, generating unquantified marine ecotoxicity impacts. In addition, the use of both mechanical and chemical delousing treatments has proven deleterious effects on farmed salmon mortality [90]. It is essential to determine how much direct and indirect impacts are linked to these various types of delousing treatments, especially when we know that in recent years, mortality represented between 6–9% of the total Norwegian production in terms of biomass [91].

Overall, inventories also suffer from limitations similar to those from the methodologies: Incomplete description and partial disclosure. While data confidentiality can explain why portions of the inventory cannot be fully disclosed, we still think process description and data reporting can be largely improved. Good research practices could consist to systematically disclose foreground systems, including the sources, time, and place of data collection. More descriptions of the production systems are also required. It would increase transparency and enable more subsequent analyses based on this data. Pelletier et al. [31] and White [59] are two good examples of how inventory data can efficiently be presented. When confidentiality is a requirement from companies, authors should work closely with the data provider to share a maximum of data while still safeguarding sensitive information. Aggregating inventory inputs can be one way to do so.

Another issue identified in this review concerns data representativity. Although most studies report collecting first-hand data for the most crucial processes (feed and grow-out), a significant fraction of the datasets are built with recycled data from previous studies. For example, the feed dataset developed in Pelletier and Tyedmers [92] and Pelletier et al. [31] was reportedly used in six subsequent salmonid LCAs, primarily because of its completeness (high representativity), availability (supplementary data), and perceived quality (highly cited paper). A few years ago, Henriksson et al. [93] suggested a protocol for the horizontal averaging of foreground and background data, accounting for data uncertainty generated by inaccurate measurement and means variability. This protocol is one of the tools LCA practitioners can use to circumvent cross-referencing of obsolete data. There is also a lack of open data on salmonid feed ingredients and the production processes of aquafeed, especially since it is estimated central for salmonid LCA to use data representative of the feed used. Substantial inclusion of an analog dataset tends to artificially increase the level of diversity and completeness (more case studies tested with similar data) of a given research area and reduces variability between studies. The construction of national inventories and open LCA databases is crucial to address these issues, and it has already been recommended in an aquaculture LCA review published so far [37,39]. We, therefore, recommend LCA practitioners to support the development of open inventories such as AGRYBALYSE [94], the Seafood LCI database [95], or the Thai LCI database [96]. We also suggest researchers to participate more actively in the GFLI (Global Feed LCA Institute) initiative and collaborate with the FEFAC (European Feed Manufacturers' Federation) to access to the latest data.

It can also be highlighted that the extensive use of case studies in salmonid LCAs (and the broader field of LCA) are not always well equipped to fully capture the environmental impacts of large production value-chains such as aquaculture. In salmonid LCA, data collection is limited in time (a production cycle) and often limited in space (one production site). Exceptions consisting of Chen et al. [28], Papatryphon et al. [24], Pelletier et al. [31], Ziegler et al. [48], and White [59] who covered significant proportions of the national productions they analyzed (global production in the case of Pelletier et al. [31]). Process-based LCA is well tailored to compare specific production parameters, sites, technology, or products, but is not optimized to achieve process completeness or to model whole sectors of an economy. Using environmental input-output models could address these issues by providing process completeness and resolution to national aquaculture sectors. Research to develop successful hybrid input-output/LCA models have been taking place for several years [97,98] and the recent development of EXIOBASE v3 [99] and its incorporations in a hybridized framework is promising [100,101]. In practice, this could be done by replacing conventional LCA background processes by data from the multi-regional supply and use input-output databases. Not only could such frameworks account for side processes usually left out of process-based LCA (e.g., office activities, bank transactions, labor), but it could also allow practitioners to integrate processes with low inclusion levels such as chemotherapeutants, equipment, infrastructure, egg production, as well as the whole upper value-chain (processing and distribution). Unsurprisingly, this method will also bring its load of challenges. Among others, LCA practitioners will likely have more difficulties to disaggregate processes, work at local or regional scales, and choose restrictive system boundaries.

4.3. Cross-study Technological Comparison

4.3.1. Outlying Results

Large intra-cluster differences observed throughout the four impact categories are due to the variable nature of production systems studied and of LCA methodology applied. Although we review similar species using standardized system boundaries and FU, LCAs are performed on specific systems, using different feed regimes, varying in time and space, and calculated with adaptive LCA methodology. This inevitably generates significant discrepancies. Yet, if we analyze the outlying results for each impact category, individual studies can be identified, and their extreme scores can be linked to methodological choices, specific inventory, or other particularities. Salmon impacts from Ayer and Tyedmers [34], White [59], and Parker [57] display consistently high outstanding impacts while d'Orbcastel et al. [53], Grönroos et al. [36], Nyhus [56], and Dekamin et al. [54] report much lower impacts for trout production. Samuel-Fitwi et al. [50] generated both fringe high and low scores depending on the impact category considered. In Ayer and Tyedmers [34] the inventory shows that both the FTAS and RAS scenario report elevated electricity consumption (13,400 kWh and 22,600 kWh, respectively). This high demand coupled to the Canadian electricity mix (61% hydro, 18% coal, 13% nuclear, 4% oil, and 4% natural gas) leads to unusually high impacts for GWP and AP (Figure 4; Figure 5). Parker [57] and White [59] studied the environmental impacts of Tasmanian salmon, commonly fed with high inclusion of animal by-products from fisheries, poultry, as well as mammals like swine and beef cattle. In these studies, the higher impacts from the feed composition are accentuated by high FCR levels (1.3 and 1.46). These factors combined explain the remarkably high GWP, AP, EP, and CED scores of these two studies. The borderline high results obtained by Samuel-Fitwi et al. [50] in GWP and AP are directly linked to the high electricity demand of the system (19,622 kWh) and the high percentage of fossil fuels in the electricity mix of the region. In fact, by replacing this mix by wind power, the RAS GWP impacts are cut by a factor of 10 [50]. The reasons for the outlaying low scores for EP and CED remain uncertain. The authors report nearly zero nutrient emissions thanks to the mechanical and biological filtration systems, yet such filters are not a unique component of this study. The significant differences between attributional and consequential LCA frameworks could explain some of these differences. Rainbow trout LCA studies appear to generate most of the extreme low values across both technology clusters and impact categories (Figures 4–7). d'Orbcastel et al. [53] reported consistently low impacts, especially for the hypothetical RAS scenario displaying a low FCR and high fish density throughout production. The hypothetical nature of this production system and the reuse of fish slurry as biofertilizer could explain these outlier results. While the corrected data provided by the author [102] somewhat leveled the differences between Grönroos et al. [36] and other studies, this assessment still reports some of the lowest scores for the closed sea-based and open sea-based clusters AP. The significant difference with the other studies remains uncertain, but we suspect the lower AP scores to be linked to outdated data (1987 to 2000), to the feed composition, which was largely based on fish meal at this time, and potentially the use of Finnish characterization factors. Similarly to d'Orbcastel et al. [53], Dekamin et al. [54] report low GWP and AP scores for the FTAS scenario and a low EP for the RAS system. In the latter, water outlet is treated by drum and bio-filters as well as ion exchange, which could explain nutrient emissions as low as 8.5 PO4-e. Based on the contribution analysis, the low GWP and AP are due to an advantageous FCR and low electricity requirement [54].

4.3.2. Cross-Study Statistical Comparison

Although a significant portion of the LCIA statistical analysis was rejected, the current data state allowed us to perform some statistical comparisons of clusters means on both LCIA and FCR data. In the LCIA four-group analysis we statistically demonstrated that the clusters mean were similar for CED, which means that there was no statistical evidence that CED of salmonids production systems was varying in pair with the production technologies. Yet, by increasing the sample size of the comparison in the two-group LCIA A-B analysis we demonstrated that sea-based and land-based production technologies correlated with CED impacts. This means that on average, sea-based outperformed land-based systems for this impact category (41,768 vs. 96,771 MJ-e). Such results have been repeatedly observed in the literature, where land-based systems generally demonstrated to have higher energy consumption due to the land-based water circulation requirements [34,36,51,60]. There is also evidence that clusters 1-2, 3-4, and 2-4 display statistically different EP scores. It means that on average closed sea-based outperform open land-based systems (26.7 kg vs. 50.6 PO4-e), closed land-based outperformed open sea-based systems (17.3 vs. 47.3 PO4-e), and closed land-based outperform open land-based systems (17.3 vs. 50.6 kg PO4-e; Figure 6). While there were strong indications that EP impacts correlate with the open and closed characteristics of productions systems, this was not strictly demonstrated since EP impacts of clusters 1-3 have equal means and the results of the two-group LCIA C-D analysis were rejected due to skewness. These results were also coherent with the scientific literature since open systems have no waste collection, and therefore emit more nutrients into the environment than their closed counterparts [42].

All the FCR data qualified for statistical analysis. This suggests that FCR data is less sensitive to some confounding factors since it is an inventory parameter and not a multivariable computed LCIA result. In the four-group FCR analysis, we demonstrated that cluster 2-3 have statistically different means while all the other cluster comparisons did not. It means that on average, open land-based outperform open sea-based systems (1.124 vs. 1.256) for the FCR. The technological characteristic determinant for this difference is identified in the two-group FCR A-B and C-D analyses, once the sample size has been expanded. The A-B comparison shows that the sea-based or land-based traits of the systems were responsible for this difference while the C-D analysis proved that the open or closed characteristic did not influence the results. Overall, it means that land-based systems have lower FCR than sea-based systems on average (1.124 vs. 1.257). We hypothesize that this significant difference is due to the more controlled environment unique to land-based facilities. At sea, salmonids are exposed to fluctuating seasonal temperatures, parasites, and latent or active disease outbreaks. These dynamic factors invariably lead to less predictable feeding, and mortality discrepancies, which directly affects the FCR of sea-systems negatively.

4.3.3. Data Quality for Meta-Analysis

One of the clear outcomes of the statistical analysis performed on LCIA and FCR data, is the bottleneck generated by the poor meta-data quality available from LCAs of salmonids. While it certainly exists other statistical protocols one could have used to compare the means of different clusters in various groups, the statistical tests we applied were selected for their robustness, simplicity and recognized comparative properties across scientific fields [103–105]. We observed that 50% of the LCIA four-group, 75% of the LCIA two-group A-B, and 100% of the LCIA two-group C-D analyses were rejected due to the skewness of the data. Small sample sizes, data skewness, variability of distributions, and the multitude of confounding factors existing between studies represent a major limitation for such statistical meta-analysis. Increasing the scope of the comparison may improve sample sizes and distributions but it will inevitably reinforce influences of confounding factors. To improve the potential of cross-study results comparison in LCA, we identify two main levers. The first one consists of significantly increasing the quantities of LCA studies available per commodities to increase sample sizes and statistical power required for more robust statistical comparisons. Today, the inertia of LCA studies are such that for certain commodities, only a few case-studies are published sporadically through time. Process-based LCA analyses are expensive to conduct and often require several months of data collection, often largely depending on the voluntary participation of companies. This bottleneck could be partially addressed using hybridized IO-LCA methodology (see 4.2. “Inventories”) to model more efficiently background systems. It could also be tackled by means of direct access to data streams from production systems, reducing drastically time and resources applied to collect data manually from companies. The quantity of data collected by companies is increasing exponentially, and the adoption of industry 4.0 (automated manufacturing using connected sensors) could drastically increase the speed and ease of access to data [106]. Such a technological revolution could also allow practitioners to perform LCA on digital twins of production systems to enable various forms of prospective environmental assessments. The second lever should provide the tools to enable full reproducibility. This implies re-thinking in-depth the current LCA methodology. Reaching full reproducibility would require major changes in LCA practices. Today, LCA reproducibility is drastically limited by data ownership, poor data availability, limited disclosure practices (see 4.1. “Methodologies” and 4.2. “Inventories”), and restrictive software licenses. To alleviate confounding factors limiting cross-study comparisons without using a strictly standardized LCA framework for all systems, practitioners need access to the full inventory in the LCA software. With direct access to the LCA models, it would be much more feasible for researchers to verify, reproduce, and modify LCAs for comparative purposes. This off-course raises issues since desegregated foreground data is usually subject to confidentiality agreements with companies, background data is subject to access to LCA databases (e.g., Ecoinvent), and most popular LCA software (e.g., SimaPro, Gabi) are licensed. A way around these challenges could consist to provide incentives for open data and open software (e.g., OpenLCA) and favorize the publications of LCAs in open access journals. Monetary data is freely open in most countries and represent one development lead. Open LCA databases are valuable initiative requiring more development (see 4.2. "Inventories"). Finally, a third alternative to democratize open data could be achieved by national manufacturing reporting schemes managed by governmental entities. For instance, in Norway, the aquaculture industry is required to report some production data (e.g., biomass produced, mortality, feed use, etc.) to the Directorate of Fisheries regularly. Such data is compiled and made available for stakeholders. With the emergence of industry 4.0, efficient data collection, and data transfer, one could imagine that authorities could collect detailed production data from companies to dynamically monitor the environmental impacts of the commodities they are producing. Emerging technologies such as distributed ledger technology could also be an integral part of such systems to certify that the data provided by the manufacturer has not been modified or tampered with [107,108].

5. Conclusions

Throughout our analyses, the idea that LCA practitioners should strive to achieve systematic reporting rather than looking for a single and standard way to apply LCA on salmonid aquaculture systems was strengthened. Using balanced levels of methodological flexibility gives researchers the freedom they need to adapt the methodology to their production system and specific goals, thereby allowing methodological innovation without compromising minimum requirements of comparability. Systematic reporting of methodological choices and inventory data is an essential comparability requirement currently lacking in the reviewed literature. We, therefore, urge LCA researchers to disclose their methodology and their foreground system inventory in their supplementary data. Such practices will facilitate future attempts to conduct reviews, but more importantly, it would allow the scientific community to access, analyze, adapt, compare, and reuse data for other purposes. Journal editors are in a position to make sure authors comply by enforcing full data and methods transparency as part of their publication requirements.

The lack of process and methodology completeness is another pitfall we identified across the literature. While sensitivity and uncertainty analyses are not required to perform an ISO-compliant LCA, these are essential research tools to measure uncertainty surrounding data, methodological choices or assumptions, and to identify the critical environmental hot spots of a system. Still, often authors omitted sensitivity and uncertainty analyses and excluded processes on uncertain basis. Although we cannot expect LCA practitioners to reach 100% completeness using process-based LCA, there is room for improvements. For instance, we expect current disease and parasite treatments performed on salmonid open sea-based systems, activities that to date have been excluded, to give significant contributions to total impacts.

Our first attempt to do a cross-study analysis based on salmonid LCA demonstrates two main results. Firstly, although the data quality remains weak, it proves possible to perform single factor comparisons across studies despite confounding factors. We were able to compare some of the data statistically and, based on differences measured, rank specific clusters according to their average EP, CED, and FCR scores. Secondly, we demonstrated that such meta-analyses are limited by the current state of LCA methods and data. To improve salmonid LCA cross-study comparison, researchers should thrive on increasing the quantity and reproducibility of LCA data and on making sure that LCA methodological developments lead towards more data availability for stakeholders. We believe that investigating the meta-analysis potential of LCA is critical for methodological progress, and more importantly, to drive sustainable development of products and services. Influencing the sustainability of production sectors through research is a challenging task for which isolated, highly specific environmental assessments are not necessarily well suited for. Industrial actors and policymakers need robust research-based information, built on overarching trends, to be able to make sustainable strategic choices for the future.

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Paper II: LCA of biological lice treatments

ERRATA

There is an error in section 2.2, "Life cycle inventory" of Paper II. Under table 1, we are stating, "we selected localities with a full production cycle comprising 104 weeks of continuous production" [...]. Instead, the sentence should be "we selected localities with a full production cycle comprising between 52 and 104 weeks of continuous production" [...].

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RESEARCH AND ANALYSIS

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Quantifying environmental impacts of cleaner fish used as sea lice treatments in salmon aquaculture with life cycle assessment

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Abstract

Increasing pressure of sea lice, development of multi-resistance to chemotherapeutants, and alternative delousing strategies have been raising concerns about the environmental impacts of salmon farming. Ectoparasitic sea lice and its treatments represent a major bottleneck for the development of the Norwegian salmonid aquaculture. The environmental impacts of different treatments and their contribution to the salmon footprint remain unknown; these processes have been excluded from life cycle assessment (LCA) of farmed salmon. In this work, we apply LCA to quantify the impacts of three different value chains expressed per ton of cleaner fish farmed/fished, distributed, and used. The impacts of farmed lumpfish, farmed wrasse, and fished wrasse are then combined to calculate the footprint of the Norwegian biological lice treatment mix, expressed per ton of salmon produced. We found that wrasse fishing generates considerably lower impacts than farmed lumpfish and, a fortiori, farmed wrasse. The direct comparison of these value chains is compromised since LCA is unable to quantify ecosystem impacts and because cleaner fish delousing efficiencies remain unknown. Overall, the impacts of biological lice treatments have a low contribution to the salmon footprint, suggesting that using this treatment type could be a sound approach to treat salmon. However, such favorable results depend on three critical factors: (1) the efficiency of biological lice treatments needs to be confirmed and quantified; (2) ecosystem impacts should be accounted for; and (3) cleaner fish welfare issues must be addressed. This article met the requirements for a gold-gold *JIE* data openness badge described at http://jie.click/badges.

KEYWORDS
cleaner fish, industrial ecology, lice treatment, life cycle assessment (LCA), salmon aquaculture, sea lice

1 | INTRODUCTION

It is essential to improve the sustainability of current food production and promote low-impact products to feed a growing population within the planetary boundaries (Conijn et al., 2018; Springmann et al., 2018). Ocean food production will be important for future food sustainability and security, but there are challenges (Costello et al., 2019). One example is the Norwegian salmon industry, aiming to increase production while respecting ambitious sustainability criteria (Meld.St.16, 2014–2015). The stagnation of production volumes reflects the severe biological conditions Norwegian farmers have been facing in recent years (Taranger et al., 2014). This is the result of intensive production of salmonids in net-pens, favoring the spread of infectious disease and parasites. Outbreaks of pancreas disease are frequent since this virus is endemic and widespread in Norway (Jansen et al., 2017). Infectious salmon anemia is also problematic, triggering isolation, and early slaughter since no vaccine or effective treatments are currently available (Hjeltnes et al., 2019). Stagnation of production is primarily due to the ectoparasitic sea lice *Lepeophtheirus salmonis* and *Caligus elongatus* and the required treatments to keep lice levels under control (Abolofia et al., 2017). Today, Norway enforces a strict lice treatment policy to protect its populations of wild salmonids, and farmers must perform delousing treatments if concentrations exceed 0.5 female lice (0.2 in the spring when juvenile wild salmon migrate to the ocean) per salmon in net-pens.

Different treatments have been used to remove sea lice: chemical, mechanical, and biological methods. The salmon industry has been employing chemical treatments for decades with chemicals administrated through feeding or bathing (Burridge et al., 2010). In 2018, emamectin benzoate was the most used delousing chemical in feeds, and hydrogen peroxide the most applied through baths (BarentsWatch, 2020). Baths are performed directly in net-pens, by crowding salmon in a tarpaulin or onboard well-boats. Mechanical treatments appeared only a few years ago through the development of processing units flushing and brushing off salmon lice. Thermal delousing was the most common mechanical treatment used in 2017, using lukewarm seawater exposure and turbulences for delousing (Overton et al., 2018). Mechanical units are usually deployed on well-boats, barges or floatable containers, and salmon are crowded in their net-pens and pumped through the processing unit. Cleaner fish feeding on sea lice are used as biological treatment. This treatment method has also been used by farmers for decades, but it is only in recent years that it gained momentum, and organized supply chains emerged. Currently, farmers are using lumpfish (*Cyclopterus lumpus*), and five species of Labridae: ballan wrasse (*Labrus bergylta*), goldsinny wrasse (*Ctenolabrus rupestris*), corkwing wrasse (*Symphodus melops*), rock cook (*Centrolabrus exoletus*) and to a lesser extend cuckoo wrasse (*Labrus mixtus*) (BarentsWatch, 2020).

In recent years, the number of treatments per ton[1] of salmon have increased and shifted from chemical to a mix of biological and mechanical methods due to lice resistance to chemotherapeutants (Aaen et al., 2015). The average use of chemical, mechanical, and biological treatments changed drastically between the periods 2012–2015 and 2016–2019, with chemical use dropping by 42% and mechanical and biological increasing by 1068% and 158%, respectively (BarentsWatch, 2020). For biological treatment, hereafter called biological lice treatments (BLT), the average deployment of cleaner fish in salmon net-pens went from approximately 14.5 to 37.5 million fish between the two time periods (BarentsWatch, 2020). The mix of cleaner fish used also evolved with the emergence of farming of the species in recent years (Helland et al., 2014; Powell et al., 2018). Increasing use of resources and shifts in lice treatment methods are raising economic and environmental concerns (Liu & Bjelland, 2014).

Environmental impacts from disease and parasites associated to food products are mostly unknown, but some research is emerging (Hospido & Sonesson, 2005; Mostert et al., 2018; Williams et al., 2015). Local ecological impacts of disease and parasites in aquaculture are well documented (Johnson et al., 2004; Kristoffersen et al., 2009; Paperna, 1991) but attempts to incorporate such impacts to the life cycle assessment (LCA) framework (with consensus) remain challenging (Bohnes & Laurent, 2018; Cao et al., 2013; Henriksson et al., 2013). Efforts to quantify the life cycle impacts of disease and parasites treatments are deficient, primarily focusing on chemotherapeutants from washing agents, antibiotics, and vaccines (Bohnes et al., 2018). So far, LCAs of salmonids produced in net pen have excluded lice treatments from their inventories (Philis et al., 2019); however, recent work underlines the importance to account for impacts of disease and parasites, including their treatments, particularly when livestock mortality and growth rates are impacted (Winther et al., 2020). Here we investigate in detail the life cycle impacts of biological delousing treatments used in Norwegian salmon farming. We quantify emissions and resource use of three alternative productions of cleaner fish as well as the BLT mix used in the Norwegian salmon aquaculture industry.

2 | METHODS

LCA is a standardized method (ISO, 2006a, 2006b) used to quantify resource use and environmental impacts. It involves mapping the inputs and outputs of biophysical flows required and generated throughout the supply chains of products and services. Conducting an LCA is an iterative process constructed around four main phases: (1) goal and scope definition, (2) life cycle inventory, (3) life cycle impact assessment, and (4) interpretation (ILCD Handbook, 2010).

[1] Throughout this study the use of "ton" refers to metric ton.

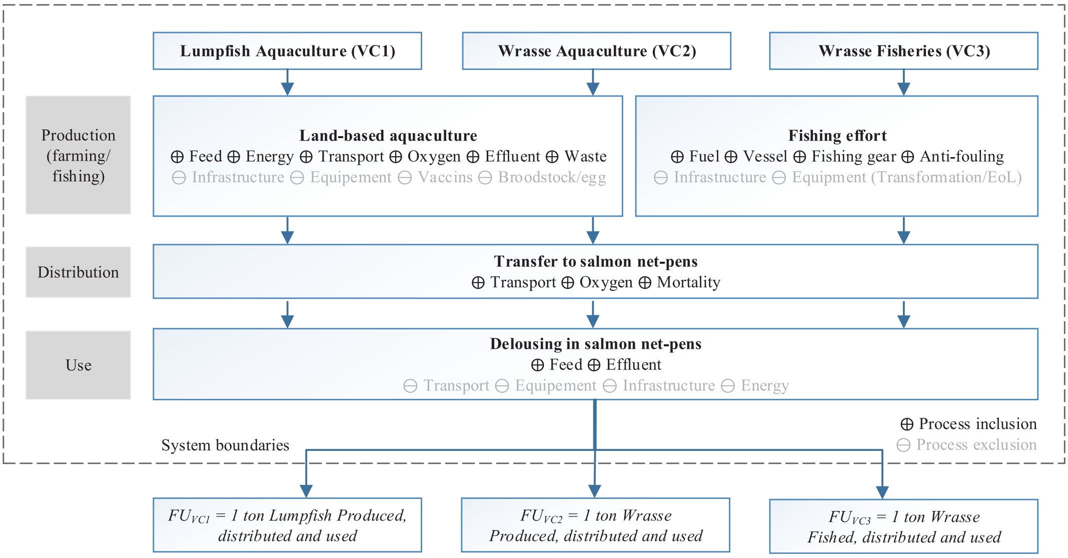

FIGURE 1 Cradle-to-grave system boundaries of VC1-3 forming the BLT used in Norwegian salmon farming

2.1 | Goal and scope definition

The objective of this LCA is dual. Firstly, the study intends to provide a comprehensive picture of impact contribution to identify hotspots in the three cleaner fish value chains (VC1-3) currently supplying Norwegian salmon farmers. Secondly, the study aims to quantify the environmental impacts of the BLT mix used in Norway and hereby evaluate how these impacts contribute to earlier LCA findings on salmon farming.

We conducted this LCA at a national scale, calculating weighted averages (based on production outputs) using representative samples of data sources. A mix of 2017–2019 data was collected from cleaner fish producers, aquafeed manufacturers, fishers, salmon farmers, national statistics, scientific literature, and LCA databases. The three value chains consist of lumpfish aquaculture (VC1), wrasse aquaculture (VC2), and wrasse fisheries (VC3) and are, with different average deployment ratio, forming BLT used by farmers to reduce the prevalence of salmon lice in net-pens. Each value chains consist of three main phases, starting with either farming or fishing, followed by distribution and use (Figure 1). Farmed lumpfish and wrasse are produced in land-based facilities, with an average rearing time of 6 and 18 months, respectively. Species also differ between wrasse value chains (VC2-3) since only ballan wrasse is farmed whereas fishers capture ballan wrasse, goldsinny wrasse, corkwing wrasse, rock cook, and cuckoo wrasse. Farmed lumpfish and fished wrasse dominate cleaner fish supply; farmed wrasse is still in an early development phase.

We selected two different Functional Units (FU) to express the environmental impacts of the value chains and of BLT. For VC1-3, the FU applied was 1 ton of cleaner fish farmed/fished, distributed, and used, following cradle-to-grave system boundaries (Figure 1). Using a functional unit based on mass was preferred to one expressed per fish since each value chains supply cleaner fish with different average weight. We converted the FU of VC1-3 into a number of fish required per ton of salmon using national statistics based on a representative population of Norwegian aquaculture sites to obtain the second FU. Both FUs are imperfect since they are not capturing the delousing efficiency, but the best options available due to lack of knowledge; comparisons between value chains and or with other treatments (mechanical, chemical) need to take this into account. Inclusion and exclusion of processes were based on estimates of expected contribution as well as data quality and availability (Figure 1). The LCA calculations were conducted in Excel and SimaPro v9.

2.2 | Life cycle inventory

Modeling farming and fishing production phases required most data collection. Farmed lumpfish and wrasse are marine species produced in land-based facilities using seawater. Out of the six cleaner fish farmers surveyed, five reported using flow-through technology and one (farming lumpfish) using a more modern recirculating system. Weighted average based on production outputs were applied to VC1-2 based on the number of producers included (VC1) or the number of production years modeled (VC2). Data from Norwegian and Swedish wrasse fishers were collected during the summers of 2018–2019. The results of the survey were divided into geographic zones starting in Sweden, near Gothenburg, moving along the Norwegian coastline, up until Nord-Trøndelag. We combined national statistics to the survey's results to scale up the processes. In 2018, the import of

TABLE 1 List of the primary data collected to model VC1-3 and BLT

Inventory type	Source	Collection	Year & representativity
Farming VC1	5 lumpfish farmers	Questionnaire, visit, calls	2017, 21% (of production)
Farming VC2	1 wrasse farmer	Questionnaire, visit, calls	2017–2018, 12–26% (of production)
Fishing VC3	61 Norwegian fishers 5 Swedish fishers Fishing statistics FDir	Phone survey Questionnaire Application to FDir	2018, 8% (of fishers) 2018, 36% (of fishers) 2018, 100% (of fishers)
Aquafeed	3 feed manufacturers	Questionnaire, calls	2017, NA
Distribution VC1-3	1 distributor	Questionnaire, calls	2018, NA
Use VC1-3	2 feed manufacturers	Questionnaire, calls	2017–2019, 100–85% (of production)
Deployment VC1-3	Treatment statistics BW for 307 localities Biomass statistics FDir for 307 localities	Downloaded from BW a Application to FDir	2017–2019, 36% (of localities) 2017–2019, 36% (of localities)

Note. VC1, lumpfish aquaculture; VC2, wrasse aquaculture; VC3, wrasse fisheries; BW, BarentsWatch; FDir, the Norwegian Directorate of Fisheries; NA, not applicable.

Swedish wrasse represented approximately 4% of the fished wrasse biomass deployed in the Norwegian net-pens. The distribution phases of VC1-3 were modeled based on the data of one distributor as well as information from the cleaner fish producers, fishers, and salmon farmers.

Aquafeed processes are necessary inputs to VC1-3 during both production and/or use phases. Cleaner fish require a variety of extruded and live feed according to their species and development stage (lice eating is a side activity for cleaner fish and not their primary source of nutrients). We modelled eight feed recipes from the feed manufacturers' data and three feed processes from open literature. For the use phase, we opted for a top-down approach, for which we gathered national lumpfish and wrasse feed sales from the main market actors. Nitrogen and phosphorus emissions to water from fish metabolism were calculated using a simple mass-balance approach based on the nutrient concentration in feeds, fish, and the collected slurry.

To calculate BLT impacts, we used records of cleaner fish deployment in salmon net-pens from BarentsWatch (BW) and the salmon biomass statistics from the Norwegian Directorate of Fisheries (FDir). We selected the BW cleaner fish deployment data for our baseline calculations over data provided by FDir since only BW data was available at the locality level. The difference of resolution and number of cleaner fish deployed reported by each institution is linked to their respective data collection methodology. BW receives the number of cleaner fish used from all the localities' production software automatically, while FDir collects the same data by asking aquaculture companies each year how many cleaner fish they have purchased and deployed. We selected localities with a full production cycle comprising 104 weeks of continuous production, between the beginning of 2017 and mid-2019. We excluded localities producing broodstock but included those using green licenses and producing ecological salmon (or trout). For each locality, the ratios of usage of the different cleaner fish were compiled, and a weighted average representative of BLT of the Norwegian production was derived. Impacts from BLT were calculated by multiplying the value chains' life cycle impacts (per fish) with the average cleaner fish mix used by Norwegian farmers.

We completed the model using Ecoinvent v3.5 (allocation, cut-off by classification), Agri-footprint v4.0 (using mass allocation), and AGRIBALYSE v1.3 (based on Ecoinvent cut-off system) LCA databases, as well as a range of assumptions. Foreground allocation based on biomass and time were conducted in VC3, focusing on the boat's construction materials and antifouling paint. A detailed description of sources, inputs, outputs, allocations and assumptions are available in the supporting information uploaded in a Zenodo repository (Philis et al., 2021). Table 1 gives an overview of the primary data collection used to model the three cleaner fish value chains and BLT.

2.3 | Life cycle impact assessment

Impact assessment was performed using ReCiPe 2016 Midpoint (H) v1.03, Cumulative Energy Demand (CED; MJ-eq) v1.11, and the AWARE Water Use (WU; m^3) v1.02 characterization methods. We selected four categories from ReCiPe: Climate Change (CC; kg CO_2 eq), Marine Eutrophication (MEU; kg N eq), Marine Ecotoxicity (MET; kg 1,4-DCB eq), and Land Use (LU; m^2a crop eq). CED is a single issue commonly used in salmonid LCAs to complete sets of midpoint impact categories (Philis et al., 2019). It measures the cumulative energy requirement of the production system, including both renewable and non-renewable sources. AWARE is a consensual water footprint method characterizing available water remaining resulting from the work of Boulay and colleagues (2018) and endorsed by the EU Join Research Center. These impact categories convert the cumulative energy and material requirements as well as emissions to air, water, and soils generated throughout the supply chains' tiers. For instance, this means that hydropower sourced electricity used to produce truck parts can generate WU and LU effects contributing to the overall transportation impacts.

We picked midpoint impacts to ensure the comparability of the results with published salmon LCAs. These six midpoint categories were selected to ensure meaningful process contribution comparisons between value chains.

2.4 | Uncertainty, sensitivity, and scenarios

Uncertainties are present from the data collection to the selection and use of characterization methods. We conducted a Monte Carlo simulation in SimaPro using 1000 iterations and a confidence interval of 95% to account for some of the uncertainty in the data. We accounted for the uncertainty generated by data variability between producers (VC1-2) and between fishing regions (VC3) using a triangular function (with a minimum and maximum) for the inputs and outputs values of the final processes (primarily for the production and fishing phase). In this LCA, baseline modeling used the Norwegian electricity production mix (including imports); this may be controversial because it varies greatly if one considers the physical or market-driven distribution of electricity. The physical approach suggests that the electricity consumed by Norwegian companies mirrors the Norwegian electricity production, i.e., almost exclusively hydropower. The market-driven perspective takes into account energy transactions between Norway and other European countries, resulting in an electricity mix much richer in fossil and nuclear sources (NVE-RME, 2019). We tested the sensitivity of results to this modeling choice by replacing the Norwegian electricity mix with the European mix. Lastly, we conducted two scenarios to investigate the effect of data choice and change in production: 1) cleaner fish deployment data from BW was replaced by numbers collected by FDir, which are reporting significantly higher cleaner fish use in net pens for the 2017–2019 period, and 2) Norwegian authorities will forbid wrasse fishing, leading to all fished wrasse being replaced by farmed wrasse on a one to one basis.

3 | RESULTS

3.1 | Life cycle inventory

The five lumpfish farmers we surveyed produced 6.5 million lumpfish in 2017. Their production volume varied from above two million to a couple hundred thousand fish per year. The modernity, technology, scale, and location of production sites differ significantly. Producers 2–5 are operating reconverted flow-through aquaculture systems, whereas producer 1 has been using a recirculating seawater system with a drum-filter and bioreactor. This farmer reported an economic Feed Conversion Ratio (eFCR) of 0.8, while others range between 1 and 1.2. Producer 3 (the northernmost farmer) used 43,000 kWh per ton of fish due to an old pumping system and more energy expenditure to warm seawater, particularly in the winter. In comparison, other farmers need between 4000 to 20,000 kWh per ton of fish. Nitrogen emissions of producers 2–5 are between 65 and 81 kg per ton of lumpfish, with variations driven by eFCR and feed composition differences; producer 1 emits around ten times less nutrients due to the recirculating technology coupled with the lowest eFCR (see Philis et al., 2021, file “SI Lumpfish Aquaculture”).

Production of wrasse in VC2 is taking place in a reconverted flow-through facility that generated 145,000 and 495,000 fish in 2017–2018. The inventory varied drastically within this period, with the eFCR decreasing from 3 to 1.1 due to a significant reduction of mortality (from disease) and major yield improvements in the second year. Following this, electricity use decreased from 209,000 to 61,500 kWh, cumulative transportation from 93,300 to 48,500 tkm, and nitrogen emissions from 180 to 54 kg per ton wrasse (see Philis et al., 2021, file “SI Wrasse Aquaculture”).

Inventory variances also occurred between wrasse fishing zones. The fuel consumption of the fishing fleet fluctuates between 800 and 1500 liters per ton of wrasse captured, with southern Norway performing best and Sweden and Møre and Romsdal worst. Vessels and gears used by fishers also differ based on location. The Swedish fleet has the highest fish to boat/trap ratio, resulting in the lowest material requirements, while fishers operating north of Trondheim use most equipment per FU. The use of antifouling paint is relatively even in Norway with values ranging between 0.8 and 1.1 kg while the Swedish fishery is using 3 to 4 times less due to its high capture rate per vessel (see Philis et al., 2021, file “SI Wrasse Fisheries”).

The distribution patterns of cleaner fish differ between and within VC1-3. In VC3, most wrasse fished south of Stavanger (southern Norway) are distributed to the salmon farmers by trucks equipped with seawater tanks. Fishers operating north of Stavanger distribute wrasse using their boat or hire the services of well-boat companies. Farmed lumpfish are distributed by trucks and well-boats, while farmed wrasse are transported by lorry only. Overall, farmed and wrasse fished in the south require the most transport. Mortality during transport was estimated at 1% for VC1-2 and 6% for VC3, based on feedbacks from the distributor and fishers. VC3 mortality is higher due to the storage period between fishing and distribution.

All feeds used have a high concentration in marine ingredients derived from fish, krill, shrimp, and squid; inclusions range from 44 to 97%, with cleaner fish feed 2 containing the least and aglonorse/nofima and otohime the most. Two feeds stand out from the others: cleaner fish feed 2 contained the highest concentration of plant-based ingredients (including Brazilian soy), and otohime included 42% krill meal (see Philis et al., 2021, file “SI Feed”). Table 2 provides an aggregated weighted average list of the inventory data collected from the producers and fishers and used to model VC1-3.

Cleaner fish deployment statistics for BLT showed that farmers used an average of 18.6 farmed lumpfish, 0.6 farmed wrasse, and 13 fished wrasse per ton of live-weight salmon produced. This average is based on all 307 localities, with some aquaculture sites using cleaner fish and some not. The

TABLE 2 VC1-3 input and output required and generated per ton of cleaner fish used (i.e., farmed/fished, distributed, and used). More detailed inventories are available in the data repository files "SI Lumpfish Aquaculture", "SI Wrasse Aquaculture", "SI Wrasse Fisheries" https://doi.org/10.5281/zenodo.4121848

Inventory	VC1	VC2	VC3
Cleaner fish (units)	24,715	29,412	23,035
Feed, pellets (kg)	1,966	1,645	248
Feed, live (kg)	5.5	217	–
Electricity (kWh)	17,128	95,076	–
Diesel (l)	14	171	525
Gasoline (l)	0.2	–	672
Transport (tkm)	15,151	58,666	10,266
Oxygen (kg)	473	957	27
Formic acid (kg)	66	38	26
Clay (kg)	–	496	–
Carbon dioxide (kg)	–	69	–
Solid waste (kg)	234	286	–
Fish silage (kg)	1,088	659	453
Nitrogen (kg)	110	102	16
Phosphorus (kg)	17	15	2.2
Vessel materials (kg)	–	–	9.5
Traps materials (kg)	–	–	58
Antifouling paint (kg)	–	–	0.5

Note. VC1, lumpfish aquaculture; VC2, wrasse aquaculture; VC3, wrasse fisheries.

minimum and maximum numbers of fish deployed are ranging between 0–185 farmed lumpfish, 0–20 farmed wrasse, and 0–168 fished wrasse per ton live-weight salmon produced (see Philis et al., 2021, file "SI Cleaner Fish Deployment").

3.2 | Life cycle impact assessment

3.2.1 | Absolute values

Environmental impacts per FU differ significantly between value chains (Figure 2). Farmed wrasse dominates impacts for CC, MET, WU, and CED while farmed lumpfish score highest on MEU and LU. Impacts of farmed wrasse are particularly high for MET, WU, and CED, with scores at least twofold larger than other value chains. Fisheries outperform farming in all categories, with particularly low impacts on MEU, MET, WU, and CED, for which it displays relative values ranging from 2 to 17% compared to the value chain with the highest score. The Monte Carlo simulation revealed varying uncertainty levels between impact categories and value chains. MET scores of VC1-2 are the most uncertain, with a coefficient of variation of approximately 41%. Uncertainty of other impacts ranges between 12–25%, except for fished wrasse MEU, with an uncertainty score close to zero. We excluded WU uncertainty due to the way water flow modeling impaired the results. The uncertainty of WU ranged between 64–126% for VC1-2, primarily because electricity-based hydropower and agricultural processes involved capture and release of water, which artificially increase the uncertainty range calculated with the Monte Carlo simulation. Multiplying the number of cleaner fish deployed by their respective value chain impacts allows to derive BLT results. These results indicate that an average ton of farmgate Norwegian salmonid generate 8.02 kg CO2 eq, 0.1 kg N eq, 0.19 kg 1,4 DCB, 1.21 m2a crop eq, 0.65 m3 and 0.18 GJ of environmental impacts due to BLT.

3.2.2 | Contribution analysis

Figure 3 displays the process contribution of the three chains. Impact distribution appears fairly homogenous for VC1-2. Feed is the largest contributor to three of the six impact categories: CC (72;44%), LU (95;68%), and WU (89;89%) for VC1 and VC2, respectively. Electricity required for farming dominates the impacts of MET (55;72%) and CED (48;69%), especially in VC2.

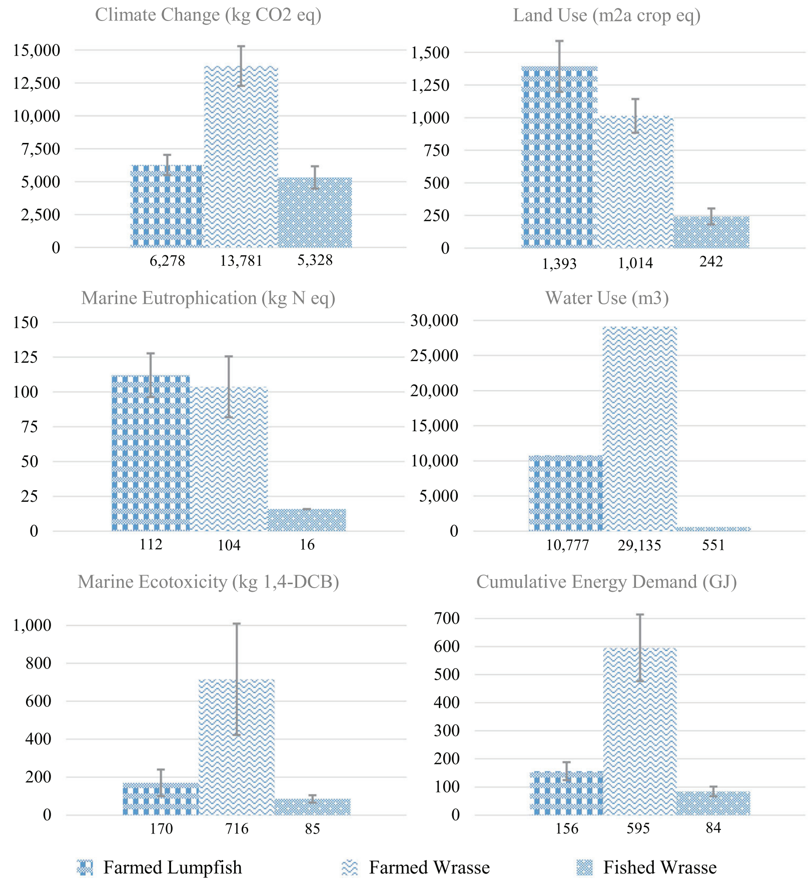

FIGURE 2 Life cycle impacts of VC1-3 per ton of cleaner fish produced/fished, distributed, and used (exact values are displayed under each graphs). Results include foreground and background inventory uncertainty estimated using Monte Carlo simulation (error bars). The elementary data used to create this graph is available on Zenodo in the repository file "SI LCIA Results" (https://doi.org/10.5281/zenodo.4121848)

Other processes like oxygen, waste management, and "remainder" (formic acid, fossil fuels) contribute between 0 to 9% to the different impact categories of VC1 and 0 to 6% of VC2. Transport remains a low contributor to VC1, with impacts ranging from 0 to 10% but account for noticeable effects on CC, LU, and CED of VC2. Wrasse fisheries' contributions differ from the two other value chains. Fuel use (both diesel and gasoline) accounts for 68% of both CC and CED and transportation contribute significantly to MET (27%), LU (28%), CED (19%), CC (17%) and WU (8%). Feed impacts are low across categories, except for LU, for which they account for 67%. The effects of vessel and gear are marginal, with only a noticeable influence on WU (30%) and MET (16%). Lastly, antifouling paint covering hulls has moderate impacts, with a noteworthy contribution to MET (36%) and WU (24%). For all three value chains, MEU impacts can almost exclusively be attributed to fish effluents. For VC1-2, emissions occur during both production and use, while for VC3, marine eutrophication only happens in the use phase (100%).

Overall, impacts of production phases are relatively homogenous across value chains, with the production phase (farming or fishing) dominating in all categories, except for MEU and LU (Figure 4). The use of cleaner fish in salmon net-pens contributes moderately to environmental impacts, except for MEU and LU of VC1 and VC3, for which it is the primary contributor. A low to moderate contribution gradient between value chains can be observed for the distribution phase, starting with VC1 exhibiting least and VC3 most impacts across categories.

3.3 | Sensitivity and scenarios

The outcome of the aquaculture value chains shows considerable sensitivity to the choice of electricity mix, particularly VC2, with the highest electricity consumption (Table 3). Replacing the Norwegian electricity mix with the European mix (which has approximately 15 times higher global

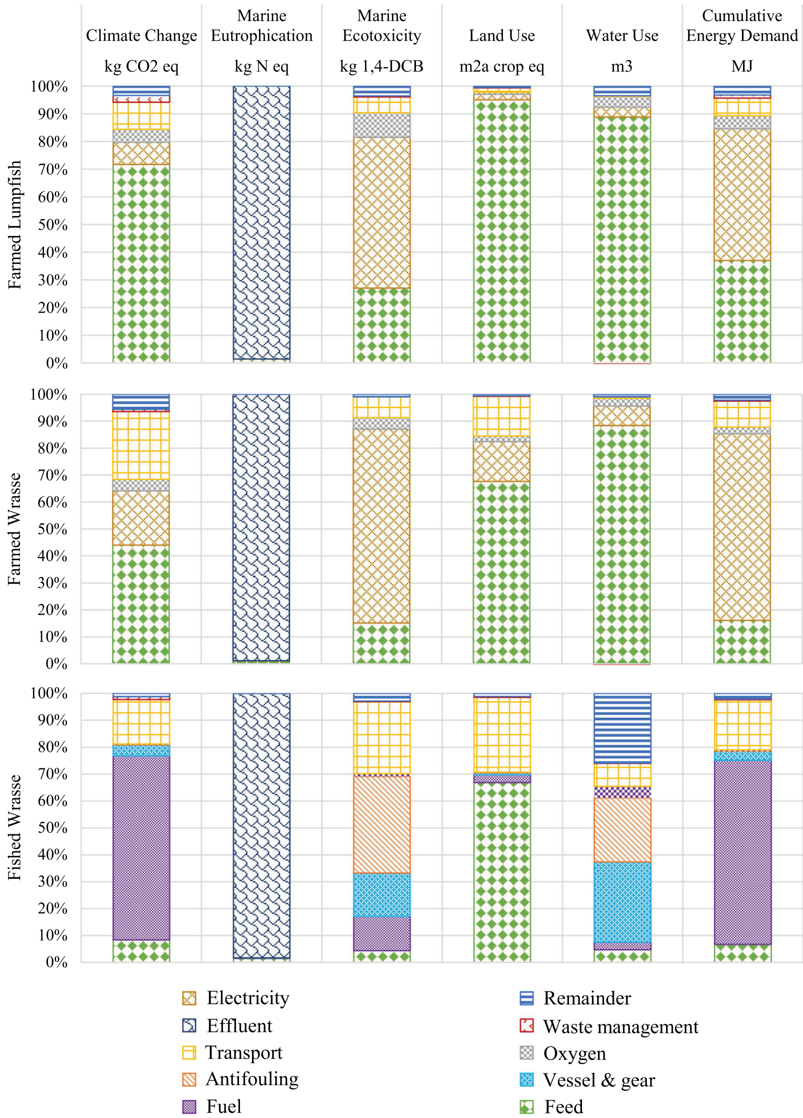

FIGURE 3 Life cycle contribution of processes used across VC1-3. Impacts are calculated per ton of cleaner fish farmed/fished, distributed, and used, and are presented in relative values. The elementary data used to create this graph is available on Zenodo in the repository file "SI LCIA Results" (https://doi.org/10.5281/zenodo.4121848)

warming potential per kWh used), modifies all VC1-2 impacts but MEU substantially. Scores drastically increase for CC, MET, LU (in VC2), CED and to a lesser extent, for LU (in VC1) and WU. On the opposite, VC3 is unaffected by the change of mix since its foreground system is electricity-free. The sensitivity of BLT to electricity mixes resembles those of VC1-2 but is dampened by the neutrality of VC3.

The higher use of cleaner fish reported by FDir (compared to BW) linearly increases deployment ratios and consequently BLT impacts. On average, FDir reports between 5 to 45% more cleaner fish deployed during the 2017–2019 period, depending on years and species. The deployment ratio of VC1-3 increase from 25–31% compared to baseline, resulting in a homogeneous rise of 26–27% of all impact categories.

Finally, a ban on wrasse fishing will affect BLT impacts, assuming VC2 would provide a simple one to one replacement. The model is very sensitive to this scenario. BLT impacts increase across all categories, especially MET (+141%), WU (+139%) and CED (+122%); it has the least effect on CC (38%).

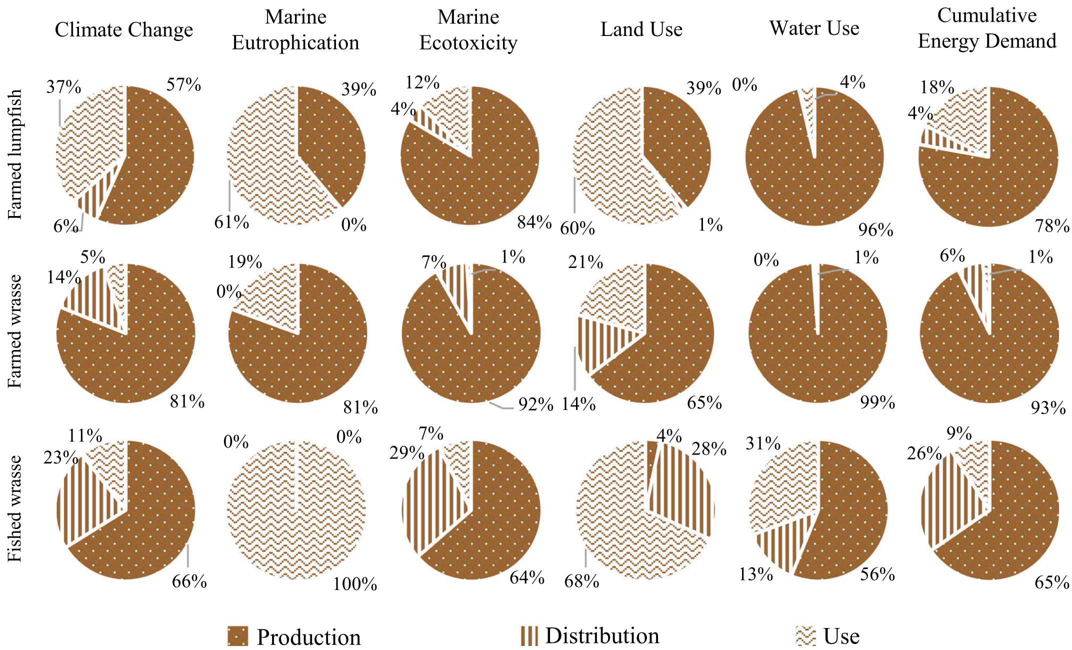

FIGURE 4 Contribution of production phases modeled across VC1-3. Impacts are calculated per ton of cleaner fish produced/fished, distributed, and used and are presented in relative values. The elementary data used to create this graph is available on Zenodo in the repository file “SI LCIA Results” (https://doi.org/10.5281/zenodo.4121848)

TABLE 3 Relative change to the baseline of VC1-3 and BLT impacts, testing sensitivity to alternative electricity mix, deployment data, and a ban of wrasse fishing. The underlying data compiled in this table is available in the repository file “SI LCIA Results” https://doi.org/10.5281/zenodo.4121848

Impact categories	VC1 EU-mix	VC2 EU-mix	VC3 EU-mix	BLT EU-mix	BLT FDir	BLT VC1-2
Climate Change	+112%	+284%	0%	+76%	+26%	+38%
Marine Eutrophication	0%	+3%	0%	0%	+27%	+39%
Marine Ecotoxicity	+171%	+226%	0%	+133%	+27%	+141%
Land Use	+17%	+129%	0%	+17%	+27%	+26%
Water Use	+11%	+24%	0%	+12	+27%	+139%
Cumulative Energy Demand	+72%	+106%	0%	+55%	+27%	+122%

Note. VC1, lumpfish aquaculture; VC2, wrasse aquaculture; VC3, wrasse fisheries; BLT, biological lice treatments; EU-mix, European electricity mix; FDir, the Norwegian Directorate of Fisheries.

4 | DISCUSSION

4.1 | Comparing the life cycle impacts of cleaner fish value chains

Strong biological differences between lumpfish and wrasse and major disparities between farming and fishing activities explain the impact variability of the three cleaner fish value chains.

Two factors are driving the large impacts generated by VC2: (1) it is the newest and smallest value chain, and (2) farming wrasse is more complex and resource-intensive compared to lumpfish. Impact reductions can be expected as VC2 grows and matures (Cucurachi et al., 2018). In fact, the data already show dramatic efficiency improvements between 2017 and 2018 (see Philis et al., 2021, file “SI Wrasse Aquaculture”). Yet, despite optimization and economies of scale, VC2 will probably maintain higher impacts than VC1 and VC3 because of its specific requirements. Wrasse farming has a production cycle three times longer than lumpfish and an unusually high electricity demand despite the use of heat-exchangers (Table 2).

This is partly due to the longer production cycle, higher sea-water temperature requirements (compared to lumpfish), and the current use of flow-through rearing technology (Brooker et al., 2018). Farmed wrasse also requires live feeding through the hatching phase and is particularly prone to disease and adaptation difficulties (Helland et al., 2014). Farmed lumpfish score only higher than farmed wrasse in two of the six impact categories: MEU and LU. The main reasons are higher feed requirements of lumpfish in the salmon net-pens, and to a lesser extent, because lumpfish producer 1 and 5 use a feed richer in agricultural ingredients (cleaner fish feed 2). VC2 shows a higher eFCR than VC1 during production (1.55 vs. 0.96), but considerably lower eFCR during the use phase (0.32 vs. 1.02), suggesting that wrasse are fed less or eat less once deployed in the salmon net-pens. Improvement options for VC1-2 would primarily consist of modernizing the production equipment by converting flow-through into recirculating systems and improving farming practices to reduce mortality and improve efficiency. Krill products are major drivers of CC and MET but replacing them with agricultural ingredients will spike MEU and LU. Overall, VC3 generates by far the lowest life cycle emissions across all categories. Impacts could even be reduced further by lowering engine power and boat sizes, as well as the distances covered during both fishing and distribution.

The sensitivity of VC1-2 to the electricity mix suggests even greater impact differences between farmed and fished cleaner fish if farmers are not supplied with abundant hydropower. Impacts will also rise dramatically across all categories if the Norwegian authorities were to ban wrasse fishing and farmed wrasse compensated the lack of cleaner fish on the market. Although lumpfish and wrasse have distinct prices and blending ratios in net-pens suggesting quantifiable disparities in delousing efficiency, the use ratios reported in localities show relative similarities. Out of 45 (VC1) and 47 (VC2-3) localities using only lumpfish or wrasse, 30 lumpfish and 25 wrasse per ton salmon produced were deployed on average; this would equate to an efficiency difference of 17% in favor of wrasse chains. Such comparison is however dubious. Localities using only lumpfish are also concentrated in northern Norway, where the salmon lice exposure is less compared to the south. Furthermore, differentiation between efficiencies of farmed and fished wrasse cannot be made with the current data.

4.2 | Comparability of delousing efficiencies

Despite our attempt to compare the environmental impacts of the different cleaner fish value chains, a significant gap of knowledge remains to couple life cycle emissions generated by the farming/fishing, distribution, and use of the cleaner fish and their potential different delousing efficiencies in the salmon net-pens. Each of the fish produced by VC1-3 has a similar function (keeping the number of female lice per salmon under 0.5), but their characteristics and efficiencies may vary widely. Differences of attributes affecting lice eating efficiencies are broad: ranging from species types, behavior, survival and adaptation rates, response to stress, growth speed, to operating sea-water temperature, and swimming abilities (Brooker et al., 2018). For instance, farmers north of Trøndelag only use lumpfish since the lower sea temperatures impair wrasse activity, but farmers further south often combine both types. Variability in biological conditions at farming sites complicates the evaluation of delousing efficiencies. Salmon lice pressure between localities depends on a multitude of factors like location, current, stock density, the prevalence of viruses, or the use of other lice treatments (Sandvik et al., 2020). There is also a lack of knowledge about the infestation levels of localities. Current measurements are done manually every week, sampling only 20 fish per cage, which represents around 0.02% of the salmon biomass.

There is a surprisingly high level of uncertainty regarding the cleaner fish delousing efficiency, with a lack of replicated studies performed at full commercial scale for each of the used species (Overton et al., 2020). Some experimental studies found lumpfish (Eliasen et al., 2018; Imsland et al., 2018) and wrasse (Leclercq et al., 2014; Skiftesvik et al., 2013) to be effective delousers. Yet, a recent top-down statistical analysis demonstrates low overall efficiency of BLT, with localities deploying cleaner fish early in their production cycle only gaining a slight delay for other treatments and thus a small reduction of the salmon lice population growth (Barrett et al., 2020).

4.3 | Cleaner fish welfare and LCA limitations

While LCA is an efficient method to account for regional and global environmental impacts generated by anthropogenic activities in value chains, it still lacks a coherent framework to account for local ecological effects and animal welfare (Ford et al., 2012; Scherer et al., 2018; Woods et al., 2016). Despite fishing quotas and seasonal restrictions established by FDir, the long-term effect of removing a large amount of wrasse from their natural environment is not fully understood (Blanco Gonzalez & de Boer, 2017; Halvorsen et al., 2017). Fishers have been reporting declining concentration of wrasse captured per traps, suggesting high fishing pressure (Skiftesvik et al., 2014). Besides, wrasse transport between fishing grounds and salmon net-pens in different counties, or even countries, is highly controversial, introducing foreign fish in ecosystems and potentially spreading disease and parasites (Faust et al., 2018; Murray, 2016).

Cleaner fish welfare is also a central issue debated among stakeholders not covered by this LCA. A recent report of the Norwegian Food Safety Authority made national headlines by reporting 40% of cleaner fish mortality during salmon production cycles (Mattilsynet, 2020). Cleaner fish mortality is particularly high in the weeks following deployment, suggesting that lumpfish and wrasse have difficulties in adapting the net-pen environments. In fact, recent cleaner fish welfare research suggests that salmon can have predatory behavior and bite cleaner fish in net-pens (Espmark et al., 2019). Wrasse seems especially sensitive to environmental stress. Inventory data of VC2-3 show abnormally low eFCR during the use phase (see Philis et al., 2021, files "SI Wrasse Aquaculture" and "SI Wrasse Fisheries). We hypothesize that lower feed inputs are unlikely to originate primarily from farmers' restriction or neglect but more from a general loss of appetite due to stress and/or, in the case of fished wrasse, the inability to feed on extruded pellets.

In reality, there is a cleaner fish mortality ratio of 100% since they are not reused at the end of each salmon production cycle. This is partly because large cleaner fish are suspected to lose their delousing efficiency, as well as disease cross-contamination risks. When salmon is ready to be slaughtered, the remaining cleaner fish are collected and euthanized, mixed with formic acid and sent to biogas facilities. Recent projects are looking for opportunities to valorize cleaner fish biomass for human consumption, pet foods, or nutrient extraction, but are still in an early development phase (FHF, 2020).

4.4 | Treatments contribution to life cycle impacts of salmon

We calculated the contribution of BLT to salmon farming by adding the treatment results to the average life cycle impacts measured per ton live-weight salmon produced in net-pens for CC and CED (Philis et al., 2019). When baseline impacts were added to the 2933 kg CO_2 eq and 38 GJ of the current salmon production estimates, BLT contributed to 0.27 and 0.47% of the new totals. With the European electricity production mix, contribution rose to 0.48 and 0.72%, and if deployment data of FDir was used, BLT accounted for 0.34 and 0.59% of the impacts. Finally, if farmed wrasse replaces fished ones, BLT will represent 0.38 and 1.03% of CC and CED, respectively.

The exclusion of infrastructure underestimates impacts from VC1-2 slightly, but since the production plants were conversions of existing buildings, their effects were estimated to be low (Bergman et al., 2020). Impacts from feed use in net-pens are also likely to be underestimated since it is known that lumpfish eat salmon pellets in addition to their feed (Eliasen et al., 2018), and inventory data indicates that wrasse are undernourished. Despite these small differences, BLT baseline and most BLT scenarios' contributions to salmon impacts remain well under any significant levels. This means that, at present the cleaner fish mix used by the Norwegian aquaculture industry has negligible effects on salmon life cycle impacts. We excluded the contributions of BLT to other impact categories due to the lack of standardized characterization methods between studies. However, results indicate that the BLT contribution is in the same order of magnitude across all six impact categories when compared to salmon. The relatively low contribution to overall emissions of BLT confirms previously reported findings based on rough estimations (Winther et al., 2020) and indicates that an increase in salmon FCR from lice infestation (due to increased mortality, reduced growth and/or salmon feed being eaten by cleaner fish) may have larger influence on salmon life cycle impacts than the impacts of BLT itself. Earlier and more recent LCA studies of farmed salmonid (e.g. Sherry & Koester, 2020; Parker, 2017; Pelletier et al., 2009; Newton & Little, 2017; Ziegler et al., 2013) did not account for sea lice treatments or other parasites and diseases affecting salmon production. Although specific to the Norwegian context, the data and methods presented here could form a basis for including this type of activities in future aquaculture LCAs.

Even if comparisons may be complicated due to lack of data on delousing efficiencies, it remains crucial to assess the current impacts of BLT with the two other lice treatments in order to give guidance for future lice management from an environmental perspective. Recent research suggests that both mechanical and chemical treatments generate significant mortality among salmon; this is likely to have larger contribution to the overall salmon life cycle impacts (Overton et al., 2018). It is also important for future research to assess the total life cycle impacts of disease, including impacts of reduced growth/increased mortality on feed use and increased need for vessel activities on the salmon farm related to treatments (Winther et al., 2020).

5 | CONCLUSIONS

Wrasse fisheries outperform aquaculture-sourced cleaner fish across the six life cycle impact categories considered. Results are strengthened if the electricity used by fish farmers is less based on hydropower. A ban or reduction of wrasse fishing due to local ecological concerns is likely to increase the life cycle environmental impacts of BLT, but not necessarily above any significance level, especially if improvements made by wrasse farmers are sustained. In order to reduce the impacts of cleaner fish farming, we recommend farmers to upgrade their production facility to a recirculating aquaculture system and use feed with less krill. Major impact reductions can also be achieved in wrasse fisheries. We suggest keeping wrasse fishing regional, using small fishing boats with low engine power, and reducing distances between capture and delivery of wrasse to salmon farmers. Relocating operations could also reduce storage time and mortality rates.

Overall, BLT impacts are low compared to the impacts of farmgate salmon and, based on the different scenarios investigated, unlikely to contribute significantly to the salmon life cycle impacts. However, the lack of evidence of cleaner fish delousing efficiency in the literature is problematic since low BLT impacts could be counter-weighted by low efficiency. This will be measurable when a FU capturing BLT delousing capabilities will be developed. Assessing the environmental impacts of biological, mechanical, and chemical treatments is essential, nonetheless. It generates the fundamental data required to add contribution of treatments to the salmon footprints and is necessary to assess which treatment mix used by salmon farmers could generate the lowest impacts. Further research is needed to quantify potentially different cleaner fish efficiency and ways to alleviate poor cleaner fish welfare currently reported in net-pens. Additional studies are necessary to quantify the life cycle impacts of mechanical and chemical lice treatments and develop more comprehensive LCA practices accounting for the direct and indirect environmental impacts of disease, parasites, and their treatments in animal-based food LCAs.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

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Paper III: LCA of sea lice treatments

This paper is awaiting publication and is not included in NTNU Open

Paper IV: LCA of warmwater fish farming in Swedish RAS

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Article

Recirculating Aquaculture Is Possible without Major Energy Tradeoff: Life Cycle Assessment of Warmwater Fish Farming in Sweden

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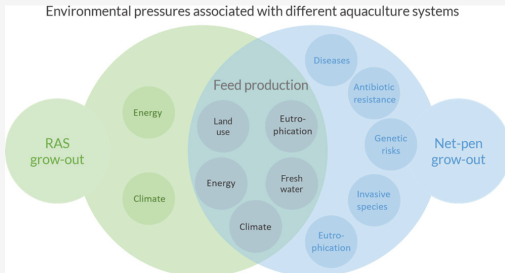
ABSTRACT: Seafood is seen as promising for more sustainable diets. The increasing production in land-based closed Recirculating Aquaculture Systems (RASs) has overcome many local environmental challenges with traditional open net-pen systems such as eutrophication. The energy needed to maintain suitable water quality, with associated emissions, has however been seen as challenging from a global perspective. This study uses Life Cycle Assessment (LCA) to investigate the environmental performance and improvement potentials of a commercial RAS farm of tilapia and Clarias in Sweden. The environmental impact categories and indicators considered were freshwater eutrophication, climate change, energy demand, land use, and dependency on animal-source feed inputs per kg of fillet. We found that feed production contributed most to all environmental impacts (between 67 and 98%) except for energy demand for tilapia, contradicting previous findings that farm-level energy use is a driver of environmental pressures. The main improvement potentials include improved by-product utilization and use of a larger proportion of plant-based feed ingredients. Together with further smaller improvement potential identified, this suggests that RASs may play a more important role in a future, environmentally sustainable food system.



■ INTRODUCTION

To achieve future food and nutrition security without jeopardizing the multiple functions of ecosystems and use resources efficiently, global food production needs to transform.[1] There are major differences in nutritional value, resource requirements, and environmental footprint between food groups and food products.[2,3] Increased understanding of the environmental performance of different production systems' and product nutritional qualities is key to ensuring human health while reducing environmental pressures. In this sense, increasing global seafood production particularly at the expense of red meat has repeatedly been identified as a promising strategy for improved sustainability.[4,5]

In European countries, dietary advice often recommends an increased consumption of seafood[6,7] based on nutritional properties. The bulk of European seafood consumption today consists of only a handful of species, with tuna dominating followed by cod, farmed salmon, Alaska pollock, and shrimp. Seafood is, however, a particularly diverse food group in terms of nutritional value and environmental footprints.[2,8,9] Around 2500 species are globally harvested from the wild and 600 species are farmed, with both systems using a wide range of production methods.[10] Some species can be produced both from fisheries and aquaculture using methods with widely different environmental impacts. Besides seafood from capture fisheries being a limited resource, there are also concerns over unsustainable fishing practices, such as for cold-water shrimp,[11] and the risk of spreading of nutrients, diseases, and parasites to the surrounding ecosystems for the current production systems (open net-pen) for farmed salmon.[12] To reduce the overall environmental impacts of the food system, these differences are important to identify and consider in national strategies and policies.

Consequently, the Swedish environmental legislation is highly restrictive in giving permits for traditional open net-pen systems due to eutrophication in coastal waters and because aquaculture production is marginal. Closed, land-based recirculating aquaculture systems (hereafter referred to as RASs) have therefore attracted interest to meet national strategies in aquaculture development as seen also in Norway,

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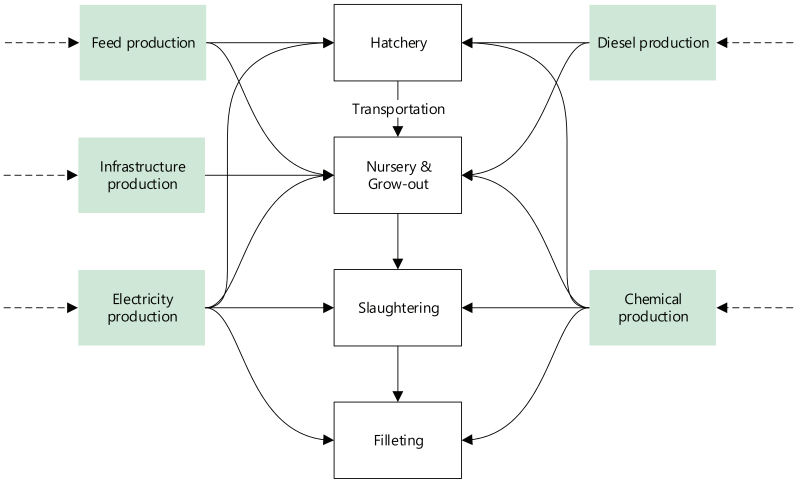

Figure 1. Simplified flowchart of the studied system with primary data in white boxes and secondary data in shaded boxes. Dashed lines indicate upstream processes.

the United States, across Europe, and China.[13,14] A RAS is generally constructed with fish tanks connected to mechanical and biological filters and with water treatment, e.g., for aeration and disinfection.[14] The treated water is recirculated back into the farming tanks while ammonia is converted into nitrate, which can be denitrified or gathered together with sludge. Being completely separated from natural ecosystems, RASs offers solutions to the main environmental issues with open net-pens as there is no risk of spreading nutrients, antimicrobials, parasites, or invasive species. Many of the recently established Swedish RAS production sites produce tropical finfish and crustacean species, such as Nile tilapia (*Oreochromis niloticus*), African Clarias catfish (*Clarias gariepinus*), and Whiteleg shrimp (*Penaeus vannamei*). These warmwater species are highly productive and require low protein inputs[15] but need to be farmed in warm water (around 30 °C).

Closed farming systems offer more controlled rearing conditions that may contribute to a lower occurrence of disease and a more efficient Feed Conversion Ratio (FCR) in the RAS than traditional open net-pen systems.[16] The improved FCR is achieved through healthier fish and better control of feeding. However, RASs require more technical input and energy to facilitate water aeration and purification in order to create suitable conditions for the fish to live and grow than in open farming systems. Previous LCAs of RASs have accordingly highlighted the environmental tradeoffs when turning to the RAS.[17,18] Ayer and Tyedmers,[17] for example, concluded that abiotic depletion (the depletion of non-renewable resources), global warming, and acidification impacts of the RAS powered by a generic Canadian electricity mix are over an order of magnitude higher than those of an open net-pen system. Song et al.,[19] similarly, concluded that electricity generation, together with feed production, dominated eight out of the nine impact categories (ranging 54–95% in total).

The goal of this study is to quantify and evaluate the current environmental performance of tilapia and Clarias produced in a commercial RAS in Sweden, as well as their improvement potential, using Life Cycle Assessment (LCA). This is one of the first LCAs evaluating a commercial RAS and the first to our knowledge to evaluate tilapia and Clarias farmed in a RAS. Increased understanding about the efficiency of these systems will help guide industry, policy, and research to further improve environmental performance and can also form a basis for strategic decisions forming a developing sector.

■ METHODS

Goal and Scope Definition. The functional unit (FU) in this study was 1 kg of fillets without skin of tilapia and Clarias (excluding packaging) following cradle-to-farmgate system boundaries. The study includes fry production, transportation of fry, grow-out in a RAS, and associated inputs (Figure 1, Table 1). The analysis covers impacts up to farmgate, which also include on-site slaughtering and hand filleting of the fish (Figure 1). Impacts associated with grow-out infrastructure and equipment were included, while the existing buildings used for farming and their maintenance were not. The business idea of the company is to use empty former farm buildings whose age (>30 years) motivates excluding their construction from the analysis. Environmental burdens were allocated among co-products (e.g., between fish meal and fish oil) based on mass as well as monetary value, with results presented for both strategies to enhance transparency and usability. Farm inputs whose use depends on space and water volume (e.g., electricity, freshwater, and tanks) were divided between tilapia and Clarias by stocking density (see the Supporting Information, Table S1) as space and freshwater are physically needed to maintain the grow-out, is related to stocking density (kg fish m^{-3}). Inputs for chemicals and equipment for water treatment were divided by fish biomass (kg) as that correlates to the volume of fish (Table S1).

Life Cycle Inventory Analysis. Data related to fish farming (nursery & grow-out, slaughtering, and filleting), representing production in 2017, were gathered directly from the largest farmer of tilapia and Clarias in Sweden. The studied system is a closed freshwater system with recirculating water in

Table 1. Farm Inputs and Outputs per Tonne Live Weight of Tilapia and Clarias Produced (Including Energy and Water for Slaughtering and Hand Filleting)

	tilapia	Clarias
economic inputs per tonne of fish		
fry (pcs)	66,768	23,276
electricity (kWh)	3086	771
diesel (l)	0.09	0.02
sodium hydroxide (kg)	0.015	0.016
sodium hypochlorite (kg)	0.30	0.31
potassium hydroxide (kg)	0.002	0.002
feed (kg)	1100	1100
hydrochloric acid, conc. 20% (kg)	0.3	0.3
transportation with truck (tkm)	6	2
plastic (kg)	2.6	2.5
iron (kg)	1.4	0.7
glass fiber plastic (kg)	4.4	1.1
environmental inputs		
freshwater (m^3)	76	19
land, grow-out site (m^2a)	33	15
economic outputs		
tilapia, live (kg)	1000	
Clarias, live (kg)		1000
environmental outputs		
N (kg)	30	30
ammonia (kg)	0.4	0.4
dinitrogen monoxide (kg)	0.7	0.7
ammonium (kg)	26	26
nitrate (kg)	11	11
P (kg)	2	2

which fish are grown in tanks connected to mechanical drum filters and moving bed biological filters to remove solids and ammonia. Electric pumps and fans were used to circulate and aerate water, and heat exchangers were used to keep water temperature around 30 °C. Fish were slaughtered in ice baths or by hand through piercing of the head. Filleting was also done manually. Energy (for heating and lighting in the combined farming and processing building and for ice production), water (for ice baths and cleaning), and cleaning agents needed for the slaughtering and filleting were not possible to separate from that for the grow-out. Inputs and outputs from the hatchery operation were based on previously published data on Chinese tilapia hatcheries,[20] adjusted to represent a Dutch hatchery by adapting the energy source and transportation by car (see the Supporting Information, Table S10 for details). Excretion of nitrogen (N) and phosphorus (P) for both farm and hatchery was calculated using a mass balance as detailed by Henriksson et al.[20] A tilapia fish feed recipe averaged over 2017 from the producer of the aquafeed for the relevant country was obtained and only adjusted to the specific levels of fishmeal and fish oil used on the studied farm. Microingredients, such as vitamins and amino acids, were excluded since detailed composition and inventory data were lacking. The electricity used on the farm was certified renewable electricity from wind, but the baseline scenario was modeled using the average Swedish electricity consumption mix as to better represent the potentials of upscaling this type of farming system rather than benchmarking this individual farm. Also, Sweden's most prevalent renewable energy source is hydropower. This is already fully utilized, suggesting that an overall increase in electricity demand needs to come from alternative energy sources. Emission estimates from such changes in demand are often assumed to come from peaking power plants, such as gas and oil, or imports, but since Sweden is expanding other renewable electricity sources, our view is that the grid average should be used.

Life Cycle Impact Assessment. Four environmental impacts were selected as relevant to assess for the aim of this study: freshwater eutrophication (ILCD 2011 Midpoint method version 1.10); climate change (IPCC 2013[21] with a timeframe of 100 years); energy demand (CED version 1.10[22]); and land use (simply estimated as the number of square meters needed annually from cradle-to-farmgate). Toxicological impact would also be of interest to assess for a food production system, but it was left out given the lack of readily available characterization factors for relevant chemicals. Acidification was excluded due to its large overlap with climate change.

Dependency on marine and poultry by-product ingredients was calculated using a slightly modified Forage Fish Dependency Ratio (FFDR) from the Aquaculture Stewardship Council (ASC) salmon standard v 1.1[23] (see the Supporting Information for further information).

All LCI modeling and characterization were performed using the SimaPro 8.5 software, with secondary data on feed ingredients from Agri-footprint (version 4.0) and the remaining secondary data from the Ecoinvent 3.4 database. Greenhouse gas (GHG) emissions from land use change (LUC) are included in the Agri-footprint data. The Agri-footprint database provides country and crop-specific emissions driven by LUC based on the PAS2050-1 framework.

Sensitivity Analysis and Alternative Scenarios. Sensitivity analyses were performed to evaluate the influence of modeling decisions and alternative farming practices to identify potential improvements. We investigated the outcome of (1) excluding LUC-associated GHG emissions, (2) utilizing entirely crop-based feed, (3) 100% utilization of filleting by-products, (4) Swedish renewable or global consumption mix as the electricity source, and (5) emitting waste nutrients to nature. When evaluating the effects of the fate of waste nutrients, no additional potential changes in the farming system (e.g., in equipment or energy demand) were considered. Crop-based feed options were based on the commercial alternative recipe for tilapia obtained from the same aquafeed producer as described above, which can maintain the same FCR according to the manufacturer.

Comparison with Other Farmed Fish. To put the environmental performance of this RAS into perspective and to bring attention to environmental tradeoffs, a comparison was made with other RAS LCAs and with fish farmed in open systems. In addition to climate change and freshwater eutrophication impacts, products were compared regarding energy consumption, fuel use, FCR, FFDR, mortality, and use of antimicrobials. Indicators were selected to capture additional relevant sustainability aspects of aquaculture. The comparison included LCAs on an early, experimental RAS production of Arctic char (*Salvelinus alpinus*),[17] a more recent large-scale RAS production of Atlantic salmon (*Salmo salar*),[19] tilapia and pangasius (*Pangasianodon hypophthalmus*) farmed in Asia in traditional ponds or cages,[20] and salmon farmed in open net-pens.[25] The two products from Asia are of the same or similar species as studied here and of relevant origin for the Swedish market,[26] whereas net-pen farmed salmon is both a different species and production system. It was included as it is

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Table 2. Life Cycle Impacts from the Production of 1 kg of Tilapia and 1 kg of Clarias Fillets Using Mass and Economic Allocation

		tilapia fillets		Clarias fillets	
impact category	unit	mass allocation	economic allocation	mass allocation	economic allocation
f. eutrophication	g P eq.	1.9	1.1	1.0	0.5
climate change	kg CO_2 eq.	14.7	7.0	9.3	4.3
land occupation	m^2a	10.2	4.5	5.9	2.2
energy demand	MJ	235	207	81	63

currently one of the most consumed species in Sweden and Europe[27] and represents aquaculture practices that RAS farming of omnivores aims to avoid (e.g., farming of carnivorous fish and farming in open systems). To enable a fair and harmonized comparison across studies using different methodologies as far as possible, climate change and eutrophication impacts were recalculated based on inventory data on the two main drivers for these systems (feed and energy). Here, the comparison stops at farmgate, disregarding edible yield, the fate of by-products, and assuming identical carbon footprints for energy and feed ingredients. While this is simplified, it avoids the strong influence of different LUC emission models and specific electricity mix, allowing a comparison of the systems conceptually, rather than the specific farms. The reported feed composition for each fish was used to calculate impacts with Agri-footprint data, and all the same assumptions and specific method choices as detailed above were used.

RESULTS AND DISCUSSION

Life Cycle Inventory Results. The studied farm produced 20 tonnes of Clarias and 20 tonnes of tilapia in 2017. The economic FCR was 1.1 for both species (Table 1). Filleting was done by hand with fillet yields of 35% for tilapia and 50% for Clarias. Filleting by-products (frames, heads, etc.) were not utilized for food or feed, only for energy production (biogas). All environmental burdens were consequently allocated to the fillets. The farm had four employees who in total worked 9600 h in the year of production. Stocking density values before slaughter were 60 and 250 kg m^{-3} for tilapia and Clarias, respectively. Pumps were almost exclusively powered by electricity, with a backup diesel generator only used on rare occasions of power failure. Chemicals were used to adjust pH and for cleaning.

Feeds are mainly constituted of plant-based ingredients, predominantly maize gluten feed, wheat, and soybean meal. In addition, 9−10% poultry by-products and 10−16% marine ingredients were included for tilapia and Clarias (see the Supporting Information, Tables S2 and S3).

Life Cycle Impact Assessment. Tilapia production was consistently associated with higher impacts across all impact categories (Table 2 and Table S4 for results per live weight), mainly as a result of the lower fillet yields and lower stocking density. The allocation strategy had a large influence on absolute values, requiring caution if comparing with other products. A major driver behind this was the large impact from feed (Figure 2) in combination with using poultry by-products that are associated with disproportionately large environmental impacts.

Despite the low FCR, feed production was the main driver behind all impact categories except energy demand for tilapia and all four impacts and indicators for Clarias (Figure 2; for economic allocation see Figures S1 and S2). Assessment of

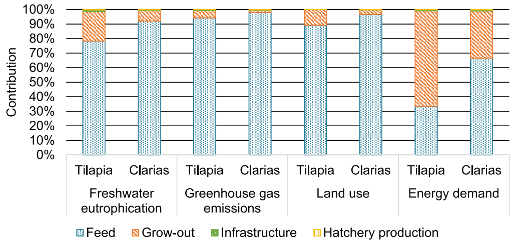

Figure 2. Life cycle contribution of 1 kg of tilapia and Clarias fillets using mass allocation.

microingredients was excluded due to the lack of data but could potentially contribute with up to 10% to climate change according to Hognes et al.[24] The dominating contribution to environmental impacts from feed has been widely observed before in LCAs of conventional production of various species.[28−31] However, it is notable that we see that same pattern for a product farmed in an RAS as previous LCA studies of such systems have shown that the extent of energy- and/or infrastructure requirements needed often overshadow the impact of feed.[19,32,33] Feed had a less dominating impact on energy demand, where the grow-out operations that include electricity use for farming accounted for 66% of the energy needed to grow tilapia and 32% for Clarias. Previous LCA studies of RASs have raised concerns about high GHG emissions related to energy provision.[17−19,33] However, the current system had a considerably lower energy use despite being located in a temperate country. Electricity consumption only varied by ±15% throughout the year while maintaining a water temperature around 30 °C in a climate that falls well below freezing in winter. Well-insulated buildings and the use of heat exchangers facilitated for efficient heat conservation. In addition, there was no need to oxygenate the systems, which can drive energy demand for the RAS.[17] In contrast to findings in LCAs of open net-pen production,[34,35] the grow-out stage of tilapia and Clarias had limited contributions to freshwater eutrophication in comparison to feed production,[30] which was expected as nutrients were retrieved and utilized. Hatcheries had only marginal contributions to overall impacts, as has been concluded for many other systems.[19,20]

Production of both tilapia and Clarias relies on animal inputs in feeds. For Clarias, the animal inputs are split equally between poultry by-products and fish inputs, while the production of tilapia relies more on poultry by-products. Clarias uses 0.48 kg of whole fish and 0.19 kg of fish by-products per kg of fillet, and tilapia uses 0.34 kg of whole fish and 0.14 kg of by-products. Adding poultry by-products, the relationship between total animal-in and fish-out in both cases is just over a one-to-one ratio (1.21 kg per kg of tilapia fillets and 1.33 for Clarias fillets).

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Table 3. Sensitivity and Scenario Analysis of Results Using Mass Allocation (Relative Change Compared to Baseline Scenario)

	freshwater eutrophication		climate change		land use		energy demand	
	tilapia	Clarias	tilapia	Clarias	tilapia	Clarias	tilapia	Clarias
by-products used	−65%	−50%	−65%	−50%	−65%	−50%	−65%	−50%
renewable electricity	−13%	−4%	−2%	−1%	−11%	−3%	−35%	−17%
global electricity mix	+350%	+109%	+91%	+25%	−7%	−2%	+21%	+10%
excluding land use change			−46%	−47%				
nutrients emitted	+312%	+401%						
vegetarian tilapia feed	−7%		−28%		−18%		−13%	

Sensitivity Analysis and Alternative Scenarios. Assumptions related to emissions from land use change strongly influenced results (Table 3; for economic allocation see the Supporting Information, Table S5) as has been shown in other LCAs of foods.[3] Excluding the GHG emissions from the land transformation, primarily driven by soy production on deforested land in Brazil, would reduce the GHG emissions from tilapia and Clarias roughly by half. This demonstrates the major improvement potential of excluding this type of soy as well as poultry fed soy from the feed.

Emitting nutrients to nature instead of using a biological filter to use nutrients as the fertilizer would result in 4 times larger eutrophication impacts for tilapia and 5 times larger for Clarias. No additional changes in the farming system were considered in the alternative scenario, but it is possible that a farming system with less water filtering would save energy. Holding sludge, however, also results in methane and nitrous oxide emissions, both powerful GHG emissions, which was not included, but could potentially give rise to considerable GHG emissions. It is therefore critical that the sludge digestion is done correctly and that the methane should ideally be collected for use as biogas.[36]

Improving by-product utilization from filleting, using more crop-based feed ingredients, and switching to renewable energy would decrease the impacts of the farmed species for all impact categories (ranked from most to least benefits). Of these, the studied farm already sources renewable energy and alternative feed sources are being investigated. Additional improvements not tested include higher fillet yields and lower FCRs. The fate of by-products strongly affects results since the fish fillets make up only 35% of tilapia's live weight and 50% of Clarias'. The by-products are currently used for biogas production, which is considered a waste treatment, thus not allocated any environmental burdens related to fish production. If by-products instead were used to produce feed ingredients and a proportion of burdens were allocated to this part, climate change impacts (mass allocated) would decrease considerably (Table 3). Pure crop-based tilapia feed would further lower impacts between 7 (freshwater eutrophication) and 28% (climate change). This feed was, however, assumed to be mainly based on soybean meal, wheat middlings, and rice bran (Figure 3). Since soy and rice productions contribute to considerable environmental impacts (e.g., from land transformation as mentioned above), it is evident that when replacing fish and poultry ingredients, attention is needed to the type of crop-based ingredient used for overall reduction of pressures. Examples of crop-based feed ingredients with a high protein content and lower climate impact are fava beans and peas (based on Ecoinvent and Agri-footprint databases). There are different ways of viewing the use of processing by-products from, e.g., poultry or fish processing. They could be viewed as free from upstream burdens, as they would potentially

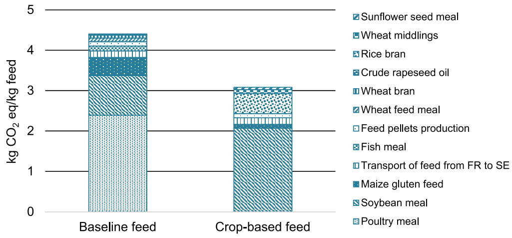

Figure 3. Greenhouse gas emissions of the two tilapia feeds.

otherwise be wasted, and as they could even replace production of another feed input. Here, we see the by-products as an integrated part of the value chain that contributes to the profitability of the main product supply chain and thereby should also share its environmental burdens.

A switch to renewable energy had limited effects on the results. This was expected since the Swedish electricity mix used in the main results is dominated by nuclear and hydropower sources. The difference in primary energy demand between energy sources is explained by energy losses accounted for in nonrenewable electricity production. The environmental footprint would increase (with the exception of land use) if the farm had been located elsewhere where electricity production is based on less renewables than it is in Sweden. Grow-out operation powered by the global electricity production/consumption mix would, for example, outsize the contribution to freshwater eutrophication for both tilapia and Clarias and slightly exceed (by 2%) the contribution of feed to GHG emissions for tilapia. This, together with lower energy consumption, contributes to differences in GHG emissions in earlier RAS LCA studies.

Land-Based Farming of Tilapia and Clarias in Comparison to Other Farmed Fish. All fed aquaculture systems have the environmental burden from feed in common, but depending on the farming system, the overall resource use, emissions, and pressures put on ecosystems are variable (Figure 4). Some of the most critical challenges concerning grow-out from net-pen farming are eutrophication and the risks of spreading invasive species, disease, or antimicrobial resistance.[12,37] Those impacts can almost be eliminated in a closed, land-based RAS but at the costs of energy and potentially climate impacts.

It is essential to acknowledge that several environmental impacts mentioned above have not been analyzed in aquaculture LCAs or cannot be assessed by the LCA methodological framework.[37] For some concerns, national statistics provide perspectives. For instance, the annual production of 1.35 million tonnes of Norwegian salmonids in net-pens (salmon and trout) generated around 160,000

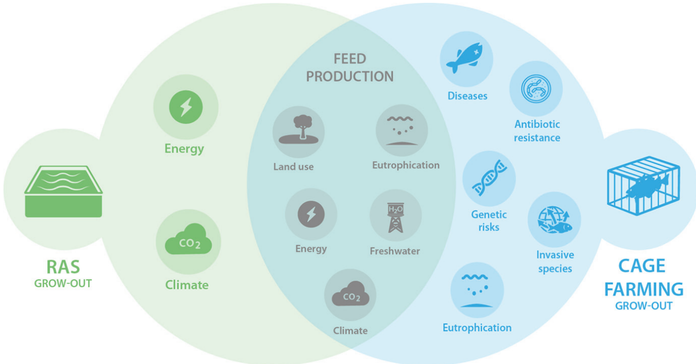

Figure 4. Environmental pressures associated with closed land-based Recirculating Aquaculture Systems (RASs) versus open systems (cage or pond aquaculture).

Table 4. Comparison of Some Performance Indicators for Farmed Fish per Tonne Live-Weight with Recalculated Eutrophication Potential and GHGe Assuming Comparable Electricity and Feed Ingredient Data

system	energy use grow-out, kWh	fuel grow-out, l	FCR	FFDR (incl. by-ps)	mortality grow-out, kg	antibiotics use, g	eutrophication, % of highest	GHGe, GLO electricity, % of highest	GHGe, SE renewable electricity, % of highest
tilapia, RAS[a]	3084	0.10	1.10	0.5	0.20	0	41%	32%	72%
Clarias, RAS[a]	771	0.02	1.10	0.7	0.25	0	32%	23%	67%
Arctic char, RAS[17]	22,600	279.00	1.45	2.2	0.30		100%	100%	100%
salmon, RAS[19]	7509	0.00	1.45	3.7	0.13		41%	42%	60%
Salmon, net-pen,[43,25,42]	0	135.00	1.32	1.9	0.05	0.1	79%	16%	52%
tilapia, ponds[20,44]	528	87.60	1.48	0.4	0.10	1.4	65%	22%	66%
pangasius, ponds[20,44]	57	1.23	1.59	0.5	0.20	93.0	75%	27%	86%

[a]This study; Ayer & Tyedmers;[17] Song et al. 2019;[19] Winther et al. 2020;[43] Ziegler et al. 2013 (mortality);[25] Henriksson et al. 2018 (antibiotics use);[42] Henriksson et al. 2015;[20] Rico et al. 2013 (antibiotics use).[44]

escapees in 2018.[38] Escapees are together with salmon lice the most potent threat to the wild salmon stocks.[39,40] Salmon lice and the treatments used to control its occurrence are putting additional pressure on ecosystems. Use of antimicrobials and other therapeutants remain an issue for aquaculture sectors including salmon farming in Chile or pangasius farming in Vietnam.[41,42] The environmental footprints of such treatments in open sea-based production are relevant to consider ensuring fair comparisons with RASs. An advantage of farming tropical species in temperate regions is that there is no risk of introducing nonindigenous species or mixing with wild fish stocks genetically. An additional benefit with freshwater species is that it allows for easy recirculation of wastes on agriculture land since wastes do not contain salt.

Differences between open and closed systems, as well as within systems and species, are shown in a comparison of seven species and production systems of farmed fish regarding both non-LCA indicators (e.g., mortality during grow-out and antimicrobial use) and two critical LCA impact categories (climate change and eutrophication) (Table 4). RAS-farmed Clarias, tilapia, and salmon had the lowest eutrophication impacts (for absolute values, see the Supporting Information, Table S6), while Arctic char farmed in the RAS was associated with the highest eutrophication potential. This was for Arctic char driven by high electricity use, also contributing to GHG emissions, while grow-out dominated eutrophying emissions from net-pen and pond systems (Table S7). Salmon in net-pen, however, had the lowest GHG emissions by assuming a global electricity mix followed by tilapia in ponds and Clarias in the RAS (Table 4). If grow-out processes instead were powered by renewable electricity, the relative environmental impact from feed increases (Table S8, Table S9) and differences between aquaculture systems and species are to some extent evened out. Antibiotics may be used in open cage and pond systems but were completely absent (or in two cases not measured) for the land-based RAS.

The RAS-farmed Clarias and tilapia had lower FCRs than all other systems. Interestingly, the FCR for RAS salmon was higher than that for salmon farmed in net-pens, which goes against previous findings that RASs generally have lower FCR.[16] This could potentially reflect differences in optimization. Tilapia and Clarias from Sweden had similar levels of forage fish dependency to tilapia and pangasius farmed in Asia (all below 1 kg of fish per kg of fish produced, meaning they are net fish producers). The three salmonid systems all rely on forage fish to a greater extent (from around 2 kg of fish per kilo of fish produced for salmon in net-pen in Norway to around 4 for salmon in the RAS in China). Other animal-based inputs into feed for the fish compared vary from 0 and 1 (Table S6).

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Edible yields differ between species. The fillet yield for tilapia in this study (35%) was considerably lower than those for Clarias (50%) and for salmon (58−88% edible).[45] The relative environmental impacts would therefore change if measured per edible yield instead of per live weight. Furthermore, the nutritional profiles differ in terms of protein and omega-3 contents.[2]

RECOMMENDATIONS

This study showed that the tradeoff between energy demand (with associated emissions) and avoiding risk to the marine environment (spreading of nutrients, disease, parasites, antimicrobial resistance, and escapees) can be smaller than previously reported for RASs. Along with the large improvement potentials observed, this suggests that RAS-farmed fish can contribute to a more sustainable food system including more seafood.

It is essential to acknowledge that many highly relevant environmental interactions for aquaculture are not possible to assess using the LCA framework, many of which RAS systems outperform open net-pen technology. This emphasizes the need to look beyond LCA results when examining sustainability of aquaculture.

Current small-scale farms could benefit from scaling-up, especially when it comes to possibilities to filleting by-products more efficiently. Up-scaling of RAS farms in Sweden would however be made easier if the environmental legislation was altered so permits were given based on environmental pressures rather than on the amount of feed used, as is the case today.

RAS systems are an emerging production technology under continuous improvement. The results here should be regarded as a snapshot of a still evolving industry in Sweden. The tilapia and Clarias RAS exhibited major improvement potentials for activities contributing most to climate change such as feed choice and utilization of by-products. Farms should focus on utilizing as much as possible fish and optimize toward using low-impact feed ingredients. Future RAS farms in Sweden are encouraged to buy specifically certified eco-labeled wind power as this most likely would increase renewable generation capacity rather than marginalizing other users. The studied RAS already shows promise, and through development in more sustainable directions identified here, land-based farming of tropical fish can contribute to a sustainable future food sector in Sweden.

ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.0c01100.

A detailed description of allocation for input data, hatchery data, and aquafeed recipes is provided in the Supporting Information. Results calculated with the alternative allocation method, functional unit, and input data are also provided (PDF)

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Notes

The authors declare no competing financial interest.

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