The handle http://hdl.handle.net/1887/36146 holds various files of this Leiden University dissertation.

**Author:** Henriksson, Patrik John Gustav  
**Title:** Evaluating European imports of Asian aquaculture products using statistically supported life cycle assessments  
**Issue Date:** 2015-11-12
Chapter 1

General Introduction

1.1 Aquaculture and food

Many of the anthropogenic pressures currently pushing the planet’s ecosystems to their limits are a direct result to food and feed production (Steffen et al. 2015): forests have to give way to agricultural fields, pastures and aquaculture ponds (Galford et al. 2010; Donato et al. 2011; Smith et al. 2013), large quantities of pesticides and herbicides are dispersed into nature (González-Rodríguez et al. 2011), enteric and other anoxic degradation of biomass result in large methane emissions (Pelletier and Tyedmers 2010a; Lindquist et al. 2012), soils are being eroded (Stoate et al. 2001; Wiloso et al. 2014), and extensive external energy inputs are needed to maintain production (Pelletier et al. 2011). As a result, biodiversity is being lost at record rates (Hooper et al. 2012; Steffen et al. 2015), natural cycles of nitrogen and phosphorus are being distorted (Bouwman et al. 2013), and the regenerative capacity of many biotic resources are being undermined by overexploitation (Foley et al. 2007; Burgess et al. 2013). Meanwhile, the planet’s human population continues to grow, as is the per capita demand for animal proteins with increasing standards of living (FAO 2006; Godfray et al. 2010).

Livestock dominates animal production by mass, but is also commonly identified as the environmentally worst food group (FAO 2006; Duarte et al. 2009; Röös et al. 2015). ‘Fish’ (see glossary), in the meantime, supplied roughly 17% of the animal proteins consumed globally in 2010 (FAO 2014a). Per capita consumption of fish has, however, doubled over the last fifty years, thanks to improved logistics, production practices and processing techniques (Muir 2005). In some parts of the world, fish are even the primary source of proteins and/or the major source of income (FAO 2014a). Capture fisheries’ catches, historically the dominating source of fish, supplied most of the increases in production until the early 1990s when landings peaked and have since stagnated, or even declined, as a result of overexploitation (FAO 2014a). Increases in demand for aquatic food products have therefore instead been met by aquaculture, the currently fastest growing animal food sector (Duarte et al. 2009). Exhibiting a rough doubling in production every decade (Duarte et al. 2009) and today providing half of all finfish consumed globally (FAO 2014a), the aquaculture industry has grown to become a cornerstone for feeding future generations.
Aquaculture holds many advantages over capture fisheries and other food production systems by avoiding undersized catches and bycatch, stabilising market prices, allowing for live fish transports and genetic improvements, and improving resilience by accommodating more diverse production practices (Muir 2005; Belton and Thilsted 2014; Troell et al. 2014). The diversity in the number of species farmed exceeds both that of agriculture (30 species make up 90% of production) and livestock production (5 species make up 90% of production), with around 35 species making up 90% of production (Duarte et al. 2009; Troell et al. 2014). Aquaculture also has the advantage of being able to shift production to meet demand and has therefore often thrived upon markets where overharvested wild fish-stocks have left shortages in supply and soaring prices (Diana 2009). The salmon industry is maybe the most notable such market, where landings of wild salmon started declining in 1990 only to be replaced by farmed salmon (Diana 2009). Similar situations exist throughout Asia, where aquaculture has maintained prices for many species at affordable levels (Belton and Thilsted 2014).

Over the more recent decades, increases in demand have shifted aquaculture production towards more intensive monoculture practices that source feed resources from globally diverse origins (Muir 2005; Tacon and Metian 2008; FAO 2014a). It is, for example, not unusual that fish today are grown on feeds containing both wild fish, agriculture products and livestock byproducts (Fig 1.1). These resources are generally opportunistically sourced from global markets, where fish may be fed raw materials originating from more than three continents, processed in another country, only to be consumed in a seafood restaurant on the opposite side of the globe.

Fig. 1.1: Modern aquaculture is a globalised industry, relying upon fisheries, agriculture and livestock products from around the world, with roughly 40% of the seafood produced entering global markets.
Intensification of production has also brought with it concerns about negative environmental consequences, including the release of nutrients and chemicals, introduction of non-indigenous species, habitat destruction, reliance on wild fish stocks, energy usage, spread/amplification of diseases and parasites, mangrove deforestation and appropriation of ecological goods and services (Beveridge et al. 1997; Pelletier and Tyedmers 2008; Henriksson et al. 2011). The increasingly diverse selection of resources used to support modern aquaculture production is also related to environmental concerns of its own. To date, most of these concerns have been related to feeds, including the provision of fishmeal, soybeans, wheat, maize, meat and bone meal, and various other resources. Other environmental concerns are related to the supply of juveniles (which still are collected in the wild for many farmed species) (Ahamed et al. 2012), provision of energy (Ayer and Tyedmers 2009), cooling/freezing (Winther et al. 2009), etc. While practices have been greatly improved over the last decade with regards to some of these concerns (Vanhonacker et al. 2011; Rico et al. 2013; Noor Uddin et al. 2013), a wide range of issues remain. Many of these issues are, however, associated to specific farming systems (e.g. eutrophication to cages systems or aquaculture ponds to deforestation). In order to identify best farming practices, one therefore needs to consider the diverse set of environmental impacts caused by fish farming, capture fisheries, agriculture, livestock farming and other supporting processes.

1.2 Aquaculture production systems

Aquaculture production can be divided into many different categories, with one of the crudest being that into farming in fresh-, brackish- or marine-water (mariculture). By weight, mariculture is dominated by aquatic plants, but by value molluscs and finfish are the main commodity groups (Table 1.1) (FAO 2014b). In freshwater, finfish makes up roughly half of aquaculture production by volume, mainly by different carp species. Representing 67% of all finfish farmed, the most common carp species include: common (Cyprinus carpio), grass (Ctenopharyngodon idella), silver (Hypophthalmichthys molitrix), bighead (Hypophthalmichthys nobilis), catla (Catla catla) and crucian (Carassius carassius) carps. Most carp species are, however, consumed locally, where they yield relatively low market prices, thus shifting the focus in monetary terms towards species like Chinese mitten crabs (Eriocheir sinensis) (6.3% by value), Nile tilapia (Oreochromis niloticus) (5.1%), whiteleg shrimps (Litopenaeus vannamei) (4.4%) and pangasius (Pangasianodon hypophthalmus) (3.3%) (Table 1.2) (FAO 2014b). While the Chinese mitten crab is a species almost solely grown and consumed in China, the other three are globally traded commodities. Frozen shrimps and finfish fillets are actually the two most frequently traded aquatic products by value, followed by fishmeal (FAO 2014b).

Table 1.1: Global production volume of aquaculture commodities in 2012 (million tonnes). From: FAO (2014b).

<table>
<thead>
<tr>
<th></th>
<th>Brackish</th>
<th>Freshwater</th>
<th>Marine</th>
<th>% of total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquatic plants</td>
<td>0.8</td>
<td>0.1</td>
<td>22.9</td>
<td>26%</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>3.2</td>
<td>2.5</td>
<td>0.7</td>
<td>7%</td>
</tr>
<tr>
<td>Finfish</td>
<td>2.1</td>
<td>37.7</td>
<td>4.4</td>
<td>49%</td>
</tr>
<tr>
<td>Molluscs</td>
<td>0.1</td>
<td>0.3</td>
<td>14.8</td>
<td>17%</td>
</tr>
<tr>
<td>Others</td>
<td>0.0</td>
<td>0.5</td>
<td>0.4</td>
<td>1%</td>
</tr>
<tr>
<td>% of total</td>
<td>7%</td>
<td>46%</td>
<td>48%</td>
<td></td>
</tr>
</tbody>
</table>
Another crude division that can be made is that into fed and non-fed aquaculture. Non-fed aquaculture refers to photosynthesisers, extensively farmed animals and filter-feeders. Filter-feeders, in turn, are mainly made up of bivalves, but also by filter-feeding finfish such as the bighead carp that are commonly found in Chinese aquaculture ponds. Extensive farming relates to systems that are large enough to maintain enough primary production to sustain the organisms farmed (Table 1.3). Fed aquaculture, on the other hand, include many of the most valued organisms and is also responsible for most of the recent increases in production. These systems can be semi-intensive, intensive or hyper-intensive, depending upon how densely they are stocked. The definitions for these different systems, however, often differ amongst publications, with the definitions used in the present research presented alongside FAO’s definitions presented in Table 1.3.

Table 1.3: Definitions for different farming intensities as presented by FAO (Crespi and Coche 2008) and in this thesis (Murray et al. 2014).

<table>
<thead>
<tr>
<th>FAO definition</th>
<th>Definition in this thesis</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extensive</td>
<td>Exclusion of predators and control of competitors yielding no more than 500 kg ha(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Improved extensive</td>
<td>n/a</td>
</tr>
<tr>
<td>Semi-intensive</td>
<td>Semi-intensive – supplementary feed and fry, yielding 0.5 to 20 tonnes ha(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Intensive</td>
<td>Intensive systems – provision of all nutritional requirements, yielding up to 200 tonnes ha(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Hyper-intensive</td>
<td>Hyper intensive – Usually pumped or gravity supplied water or cage-based, yielding more than 200 tonnes ha(^{-1}) yr(^{-1})</td>
</tr>
</tbody>
</table>

Table 1.2: Global value of aquaculture commodities in 2012 (million USD). From: FAO (2014b).
1.3 Problem identification

Like the production of most commodities in today’s globalised society, modern aquaculture subjects stress on ecosystems around the world. For example, half of the fishmeal currently used in fed aquaculture practices originates from the Peruvian anchoveta (*Engraulis ringens*) fishery, a fishery that already has experienced a collapse due to overfishing (Sandweiss *et al.* 2004). Soybean is another resource commonly used in aquaculture feeds that is related to its own controversies, with marginal demands subjecting the Amazon forest to constant encroachments by agricultural farmers seeking new fertile agricultural soils (Dalgaard 2008; Galford *et al.* 2010). Similar problems apply to cattle and other livestock production systems that displace large land areas either as pastures or for feed provision (Cederberg *et al.* 2011; Middelaar *et al.* 2013). More global concerns involve the great dependence on inorganic fertilisers in agricultural practices that threatens to unbalance global nutrient cycles and exhaust finite resource deposits, while consuming vast quantities of energy (Rockström *et al.* 2009; Cordell *et al.* 2009; Pelletier and Leip 2013). Food production actually accounts for 20% to 25% of the energy use in developed countries, with an average of about five kcal of anthropogenic energy (mainly fossil fuels) going into each kcal of food produced (Carlsson-Kanyama 2003). In addition to this, it is estimated that about 30% to 40% of global food supplies never are consumed, but end up as waste (Godfray *et al.* 2010); a fraction that might be higher for fresh fish as it is a highly perishable commodity.

Apart from reducing food waste, the most efficient way of shrinking the environmental footprint of food provision is to promote more sustainable food products, while trying to displace any detrimental hot-spots in the production chain. In order to identify these hot-spots, the whole value chain needs to be evaluated. Life cycle assessment (LCA) is a quantitative tool developed to perform such evaluations. The tool has a history running back more than forty years and has over the last decades become commonplace in environmental standards, labelling schemes, policy and even legislation (Guinée *et al.* 2011). Supported by its own ISO standard (ISO 14044 2006), LCA is often said to capture a product’s environmental impacts from “cradle to grave”. Thus referring to emissions resulting from the extraction of raw materials, to the end of life of those materials. The most common goal when applying LCAs, however, is to determine if product A is environmentally more sound than product B (Guinée *et al.* 2011).

Comparisons of LCA results have, to date, mainly been done on a point-value basis. In the meantime, there are many discrepancies influencing outcomes, originating from methodological choices and sourcing of data (de Koning *et al.* 2009). In response, several initiatives have tried to standardise methodological choices (e.g. JRC’s ILCD handbook or UNEP-SETAC’s Life Cycle Initiative) and a number of LCA studies have quantified the uncertainties around LCA results. Limited consensus has, however, been reached with regards to methodological choices, as the goals of studies differ, as does the mindsets of practitioners. Quantified uncertainty estimates have also had limited success as they generally have been too data intensive or complex, restraining practitioners to only consider some sources of uncertainty or regress to conjectural estimates. Most LCA studies also lack a predefined hypothesis, indicating a rare use of significance tests in the field of LCA. Little is therefore known about the level of confidence behind LCA conclusions (Huijbregts *et al.* 2004).
1.4 Research questions

The objective of the present research was to evaluate European imports of Asian aquaculture products using LCA. The main research question was accordingly:

Are there significant differences among the environmental impacts resulting from the production of Asian aquaculture commodities, and if so, what are the main causes?

In order to address this question, four sub-question arose and were addressed:

1. Are there shortcomings in methods, data or coverage in existing aquaculture LCAs?
2. Can variance parameters be defined for unit process data in aquaculture LCAs?
3. Can these variance parameters be processed into ranges of results?
4. How can we determine if the LCA results of two systems fulfilling the same functional unit are significantly different?

In order to address the main research question, a total of 21 LCAs were conducted for four major aquaculture commodities commonly found in European freezers, namely frozen Pangasius fillets, tilapia fillets, peeled tail-on (PTO) shrimp and headless shell-on (HLSO) prawn from Bangladesh, China, Thailand and Vietnam.

1.5 The species and countries under study

Asia has always been dominating when it comes to aquaculture production and still accounts for (88%) of global production by weight (FAO 2014a). China is the main producing country, but also a major consumer, of aquaculture products. Europe, in turn, is struggling with declining fisheries landings, while being the origin of only 4% of global aquaculture production (FAO 2014b). Europe is, in the meantime, home to some of the world’s largest fish consuming nations, including Spain, Portugal and Norway, all which have an annual per capita consumption of over 40 kg. This has resulted in Europe becoming the largest single market for international trade of aquatic products, responsible for 36% of total world imports by value (FAO 2014a).

The selection of the cultured species investigated in the current research was based upon their long export history, large trade or rapid growth. The two shrimp species are often farmed under similar circumstances in brackish water (Fig. 1.2), with the indigenous Asian tiger shrimp (*Panaeus monodon*) being replaced by the whiteleg shrimp (*Litopenaeus vannamei*) from the Eastern Pacific due to persisting disease problems (Lebel *et al.* 2010). In contrast to these two crustaceans, the indigenous giant river prawn (*Macrobrachium rosenbergii*) was selected as a crustacean representative produced in freshwater systems, where production practices have evolved in response to local opportunities and resource constraints, rather than to global demands (Fig. 1.3). Two freshwater finfish were also evaluated, namely Pangasius catfish (Fig. 1.4) and tilapia (Fig. 1.5). Tilapia is the common name for a wide range of cichlids originating from Africa that have become extremely popular in Asian aquaculture, with the Nile tilapia (*Oreochromis niloticus*) being the most commonly farmed species (71% of Asian tilapia production by weight) (FAO 2014b). However, many other species of tilapia are prevalent in Asian aquaculture, including countless strains of
hybrids (Thodesen et al. 2013). Pangasius catfish (Pangasius spp.), in the meantime, constitute a more closely related group of freshwater finfish indigenous to South East Asia. The striped catfish (Pangasius hypophthalmus) is the most commonly farmed species and the Mekong delta the dominating producing region.

Unlike salmon that is a carnivorous species, all of the species here under study are omnivorous, allowing for lower inclusions of fishmeal and fish oil in diets, and therefore potentially greater net gains of fish protein. In the meantime, many Asian countries lack regulations on farming practices, chemical use, employment conditions, water treatment, etc., that are often expected by European consumers. In addition to this, many aquaculture systems in Asia are reliant on local resources that are related to their own sets of environmental interactions. For example, fish are often fed rice derived products (rice bran, rice husks, boiled rice, etc.), a crop that is responsible for a large share of anthropogenic methane emissions (Yan et al. 2009). There is also a flow of regionally caught low-value fish (also referred to as trash fish) into Asian aquaculture production, with social and environmental consequences (Edwards et al. 2004; Cao et al. 2015). Nutrient run-off from aquaculture cages and ponds have also resulted in regional algae blooms and anoxic aquatic conditions (Verdegem 2013). This in addition to the many concerns raised above highlights the broad range of both proximal and global environmental impacts related to Asian aquaculture production.
In order to evaluate the environmental impacts throughout production chains in a systematic way, the present research applied LCA to different Asian aquaculture production systems. Given the diversity of producing countries and production systems, a selection limited the scope of the study to four countries, five species and 21 production systems. The environmental impacts evaluated were also limited to those supported by rigorous impact assessment methods, deemed relevant to aquaculture production and relevant to the inventory data.

1.6 Life Cycle Assessment (LCA) and associated uncertainties

The ISO standard (ISO 14040 2006) define four phases of an LCA: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA) and interpretation. The goal and scope is a qualitative description of the methodological choices and assumptions made throughout the LCA. These choices are important as they greatly influence outcomes, which is why several guidelines have been produced to harmonise results (ISO 14044 2006; BSI 2008; JRC 2010a; JRC 2010b). The second phase, the LCI, is the most data intensive part of most LCAs, detailing all the connections among economic and environmental flows entering or exiting the product’s lifecycle. Several software and databases have in response been established to make LCIs more easily attainable and complete, ecoinvent being the most extensive. In the LCIA phase, the environmental flows from the LCI are classified and characterised towards the impact categories detailed in the goal and scope. Each impact category is generally supported by one or more impact assessment methods that can be either midpoint-oriented (characterising elementary flows to a common indicator close to the elementary flows, e.g. radiative forcing for greenhouse gas emissions) or endpoint-oriented (characterising elementary flows to a common indicator close to the areas of protection, e.g. temperature increase for the same greenhouse gases). Finally the outcomes are evaluated and conclusions are drawn in the interpretation phase.

LCA results have, to date, generally been presented as point-values that are often compared to each other without any indication of the confidence behind the estimates. In the meantime, methodological choices and assumptions made in the goal and scope can have huge influence on outcomes (de Koning et al. 2009). For studies following the same standard these discrepancies can, at least in theory, be greatly reduced. This is, however, more difficult in the LCI phase, as each of the diverse production systems supporting aquaculture farming are subject to their own sets of uncertainties and variability (from here on jointly referred to as dispersions). Ranging from natural fluctuations in fish stocks (Sandweiss et al. 2004), to variations in agricultural yields (Naylor et al. 1997), to uncertainty around the emissions from manure management (De Vries et al. 2013), to simply different energy efficiencies in machinery, these variables are next to impossible to normalise across studies.

Also in the LCIA phase are there uncertainties related to the classification and characterisation of environmental emissions. All these different sources of dispersions have, up until recently, mainly been addressed by performing sensitivity analyses (Lloyd and Ries 2007). While sensitivity analyses are useful for increasing the understanding of the relationships between input parameters/choices and results (Middelaar et al. 2013; van der Harst and Potting 2014), they fail to account for the cumulative effect from all dispersions influencing aggregated LCA results and therefore greatly limit available post-hoc analyses. A more holistic indicator of the accuracy of LCA results can
instead be generated by quantifying and aggregating the dispersions related to input parameters and choices.

The importance of providing quantified dispersions around LCA results have since long been recognised (Hanssen and Asbjørnsen 1996; Finnveden 1998) and repeatedly upturned (Björklund 2002; Ross et al. 2002). Ross et al. (2002), for example, state: “If practitioners of LCA continue to neglect the problem of uncertainty in their work, they run the risk of generating conclusions that cannot be justified by the indicator results”. In response, early estimates of inherent uncertainties were also quantified as early as in the 1990s for a number of emission parameters (Hanssen and Asbjørnsen 1996; Finnveden 1998). Around the same time, there were also several new methodologies suggested for how to include quantitative uncertainties in LCIs (Heijungs 1996; Weidema and Wesnaes 1996; Huijbregts 1998a; Huijbregts 1998b; Huijbregts et al. 2001). Weidema and Wesnaes (1996), for example, presented a pedigree approach for addressing data quality issues in LCIs, while Heijungs (1996) suggested a more empirical approach. These and other efforts were followed up by Huijbregts (2001), who also developed the ideas surrounding uncertainties related to characterisation factors. Despite these initiatives, only a few LCA studies had quantified uncertainties around results at the beginning of the present research (Lloyd and Ries 2007), none of which focused on aquaculture production. Most of the studies that did quantify uncertainties, moreover, only evaluated specific sources of uncertainty and/or only used conjectural dispersion estimates. In other cases, the drivers behind the presented ranges simply remained unclear (Steinmann et al. 2014). Concerns were even raised that if all dispersion sources were accounted for, LCA results might be rendered meaningless (Huijbregts et al. 2004).

Only in the past few years have some LCA studies moved closer towards a complete inclusion of dispersions derived from empirical data (Mattila et al. 2011; Steinmann et al. 2014; Hauck et al. 2014). The reasons for why quantitative uncertainties have not been more extensively implemented before are many, including lack of data, no uncertainty estimates in databases, many unquantifiable sources of dispersions, an absence of propagation methods in LCA software, insufficient computing power, time limitations, or simply the lack of a comprehensive methodology (Björklund 2002; Ross et al. 2002; Lloyd and Ries 2007). For example, data limitations forced ecoinvent (v2) to only rely upon generic uncertainties and a pedigree approach when they finally included uncertainty estimates in their LCI database (Frischknecht et al. 2007b). This sudden widespread availability of uncertainty parameters, however, initiated many software developers to allow for the inclusion of uncertainties. In the meantime, computing power and software algorithms improved substantially, providing the standard personal laptop with more than sufficient processing power for normal dispersion calculations. Remaining unresolved, however, was a method that allowed for the inclusion of quantified dispersions based upon empirical data together with a standardised nomenclature. While some attempts had been made to meet this need, their outcomes were often too complex to be attainable to LCA modellers, which, in their defence, already need to be experts in two topics apart from statistical theory (those of LCA and the production system under study).

By developing a method for identifying and quantifying the dispersions around LCA results, the quantification of dispersions around results could become commonplace. This would also allow for the implementation of significance tests, which, in turn, would allow for statistically supported conclusions to be made. This would further open questions about the type of significance tests to be
used, with regards to the characteristics of the data. Applying the wrong test might result in Type I statistical error, where a null-hypothesis is falsely rejected (false positive), or a Type II statistical error, where the null-hypothesis is falsely retained (false negative). In addition, being transparent about eventual shortcomings of analyses made is of utter importance, especially with regards to LCA that is an applied science where many values are ‘soft’ and underpinned by subjective judgement (Ravetz 1999).

1.7 Thesis outline

In order to provide a platform to extend this research from, Chapter 2 of this thesis presents a review of existing aquaculture LCAs (Henriksson et al. 2012c). At the time of the review, twelve peer-reviewed LCAs of aquaculture systems were found in the literature, two of which were PhD theses. The LCA studies were evaluated on the systems evaluated, methodological choices made, data sourcing, interpretation techniques and conclusions drawn.

From the review, data sourcing and data quality stood out as important topics for improving aquaculture LCAs. This is also the topic of Chapter 3 of the present thesis — A protocol for horizontal averaging of unit process data—including estimates for uncertainty (Henriksson et al. 2013). Building upon earlier efforts by Funtowicz and Ravetz (1990), Heijungs (1996), Weidema and Wesnaes (1996), Huijbregts (2001), Sonnemann et al. (2011) and others, this article tries to identify the major sources of dispersions in unit process data and present a workable method for quantifying these (Henriksson et al. 2013).

In order to evaluate the protocol and to identify the best method for propagating unit processes into LCI results, an updated unit process dataset for coal-based energy in China was used as an example. Chinese coal power was selected as it presents a much more limited model than the generally diverse aquaculture production chains, and since it was surprisingly poorly represented in LCA literature and inventory databases. Chapter 4 consequently explores different levels for averaging unit process data and methods for propagating these into results (Henriksson et al. 2014c).

Once ranges could be produced as results, the question of which conclusions could be drawn from these arose. In Chapter 5, entitled “Product carbon footprints and their uncertainties in comparative decision contexts”, an approach for propagating and interpreting LCA results using significance tests was therefore developed (Henriksson et al. 2015a). This chapter also highlighted the importance of defining a hypothesis to work towards.

In Chapter 6, the methodological advancements developed were finally used to test the main research question. Using significance tests, the hypothesis “different production systems providing the same aquaculture commodity to European consumers are associated with different environmental impacts” was tested (Henriksson et al. 2015b). Three impact categories (global warming, eutrophication and freshwater ecotoxicity) were evaluated. Alongside identifying production systems associated with significantly lower environmental impacts, best practices are promoted.