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Domestic or imported? An assessment of carbon footprints and sustainability of seafood consumed in Australia

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ABSTRACT

The distance between where food is produced and consumed is increasing, and is often taken as evidence of an unsustainable global food system. Seafood is a highly traded commodity yet seafood sustainability assessments do not typically consider the impacts of the movement of products beyond the fishery or farm. Here we use life cycle assessment to examine the carbon footprint of the production and distribution of select seafood products that are consumed in Australia and determine differences in the sustainability of imports and their domestically produced counterparts. We found that the distance food is transported is not the main determinant of food sustainability. Despite the increased distance between production and consumption, carbon footprints of meals from imported seafood are similar to meals consisting of domestically produced seafood, and sometimes lower, depending on the seafood consumed. In combining LCA with existing seafood sustainability criteria the trade-offs between sustainability targets become more apparent. Carbon 'footprinting' is one metric that can be incorporated in assessments of sustainability, thereby demonstrating a broader perspective of the environmental cost of food production and consumption.

1. Introduction

Global food trade is increasing at a faster rate than food production (Ercsey-Ravasz et al., 2012) and population growth (FAOSTAT/Tradestat, 2009) and the distances between production and consumption are rapidly increasing (Thomas et al., 2014; Watson et al., 2015). Global supply chains place great demands on ecosystems and natural resources (Tilman and Clark, 2014; Wible et al., 2014) and localised food systems have been promoted within academic literature, public policy and alternative food movements as a more sustainable option (Hendrickson et al., 2002; La Trobe and Acott, 2000; Lang and Heasman, 2009; Legislative Assembly of Ontario, 2013). Trends in trade of fish and fishery products run counter to aspirations of localised production as they are some of the most-traded food commodities worldwide (FAO, 2014), with the world's major importers, the United States of America (USA) and Japan dependent on imports for about 60% and 54%, respectively, of their seafood consumption (FAO, 2012).

Compared to agriculture, fisheries are poorly represented in food policy (Lang and Heasman, 2009) and sustainable seafood policies are being developed in isolation from other food policy. Conventionally, seafood sustainability has tended to be focused on issues concerning the harvesting of fish as a natural resource (Olson et al., 2014) and as a result, management of sustainability within capture fisheries is concerned with ecological issues such as overfishing, stock biomass and recruitment, and in some more complex management regimes, ecosystem impacts and bycatch through an ecosystem-based fishery management (EBFM) approach (Hilborn et al., 2015; Zhou et al., 2010). Similarly, management of sustainability in aquaculture systems is largely concerned with production issues including impacts of invasive species on local biodiversity (Silva et al., 2009), disease control (Bondad-Reantaso et al., 2005), impacts of chemical use on environmental and human health (Burridge et al., 2010), eutrophication of natural waterways, sensitive land conversion, and the

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use of wild fish in feed (Cao et al., 2015; Diana, 2009; Naylor et al., 2000). Consideration of the broader supply chain impacts of seafood supply is relatively recent (Avadí and Fréon, 2013; Henriksson et al., 2012; Parker, 2012).

Rising greenhouse-gas emissions are affecting food production from the land and sea (Campbell, 2014; IPCC, 2014) and the supply of seafood contributes to these rising emissions (Tyedmers et al., 2005). Achieving a more holistic determination of seafood sustainability requires consideration of emissions generated along seafood supply chains, such as product carbon footprints, as well as traditional measures of sustainability at capture or culture. Australia provides an interesting case study for examining different sustainability measures, and the compatibilities or trade-offs that emerge between them. Australia has been ranked in the top five countries for fisheries management (Pitcher et al., 2009) and the majority of commercial fish stocks in Australia have been assessed as sustainable (Woodhams et al., 2013). However, nearly 72% of the seafood consumed in Australia is imported (Ruello, 2011) and growth in consumption of imports is expected to continue into the future, in line with government food frameworks (DAFF, 2013) and to meet consumer demand for low-cost seafood products (Department of Agriculture, 2013).

This paper quantifies an aspect of sustainability that is not typically assessed in the production and distribution of select seafood products available in Australia, the carbon footprint (CF). We use life cycle assessment (LCA) to compare the CF of three domestic wild-capture products with imports that are readily substituted by consumers. We identify patterns in the emissions of different species, production methods and supply chain stages, and examine these results in the context of existing seafood sustainability assessments. We also identify the trade-offs and opportunities in combining LCA with existing seafood sustainability criteria and discuss the need for broader assessments to operationalise holistic, system-wide concepts of food sustainability and inform emerging food policy, in particular in terms of reducing carbon emissions.

2. Methods

2.1. Australian seafood imports

Australia's seafood imports consist mainly of lower-value products such as frozen fillets, frozen prawns (where 'prawns' refers to both shrimp and prawn within Caridea and Dendrobranchiata) and canned fish (Department of Agriculture, 2013). Frozen and thawed catfish (*Pangasius*) fillets from farms in Vietnam are now the most commonly eaten import (Ruello, 2011). A small amount of high value products such as lobster and abalone are also imported. The four most important sources of seafood imports to Australia are Thailand, New Zealand, Vietnam and China (Ruello, 2011), however, prawn, fish and lobster imports are sourced from around 100 different countries (ABARES, 2012).

Most seafood imported into Australia is sent by ship with approximately 10% sent by airfreight. Almost all annual imports of prepared or preserved prawns are sent by sea (ABARES, 2012). In contrast, some products such as fresh or chilled fish fillets (Australian Customs Service statistical code 304100042) are mostly airfreighted. The majority of lobster imported into Australia is frozen and transported by sea. Small volumes of fresh lobsters are flown to Australia from South East Asia and New Zealand, some of which are re-imports which have been caught in Australia and sent overseas for processing.

Data purchased from ABS was used to calculate volume, country of origin and transport mode for several product categories of imported prawn, fish and lobster over the past 10 years (www.abs.gov.au). While no data was available for some product groups, 82% of imports were included in this study.

2.2. Life cycle assessment

LCA is an integrated tool for quantifying and comparing potential environmental impacts throughout the life cycle of a product or products. The methods used in LCA are standardised through the International Organization for Standardization (ISO, 2006). In this study we compare results from LCAs on four select Australian fisheries: Tasmanian southern rock lobster (*Jasus edwardsii*), white banana prawn (*Fenneropenaeus merguiensis*) from the Northern Prawn Fishery, Australian salmon (*Arripis trutta*) fished in Tasmania and flathead (*Neoplatycephalus richardsoni*) from the Commonwealth Trawl Fishery (CTS), with products included on the Australian Bureau of Statistics (ABS) list of imports, documented in six peer-reviewed LCAs, one conference paper and two PhD theses (see supplementary material Tables S1.1, S1.4 and S1.5). These studies cover three of the five most consumed seafood groups in Australia, including prawns, fish consumed crumbed/battered – which is predominantly imported catfish (Ruello, 2011) – and Atlantic salmon (Danenberg et al., 2012), as well as a luxury seafood and several less popular fish species.

The LCA was modelled using SimaPro Software version 7.1.6. with the impact assessment method CML-IA baseline, developed by the Center of Environmental Science (CML) of Leiden University (Universiteit Leiden, 2015). All studies included use the same data libraries and LCA impact assessment method as recommended by (Baumann and Tillman, 2004). To ensure maximum comparability between studies we focused on one impact category (Henriksson et al., 2015), the global warming potential (GWP), based on the characterisation model developed by the Intergovernmental Panel on Climate Change (IPCC), where the GWP for a time horizon of 100 years (GWP100) is expressed in kilograms of carbon dioxide equivalent. We assume the CF to be equivalent to GWP, where both are measured in units of CO_2e. The functional unit (FU) for all products is 1 kg of whole product. Where transport is included, the FU is 1 kg frozen product when transported by sea. For canned Atlantic salmon, the FU is 1 kg whole fish and the transport method is seafreight, but energy use for the refrigerated container is not included. For wild-capture Australian prawns the FU is whole frozen product and for southern rock lobster the FU is live product.

All studies employed mass allocation. Published LCAs using other allocation methods were excluded from analysis. Original data was collected for the Australian LCAs and sourced from Ecoinvent libraries where not otherwise available. The system boundary for all wild-capture studies included fuel, gear and bait up to the point of landing but excluded infrastructure. Sensitivity analysis was performed where variation existed between studies regarding the inclusion of refrigeration and refrigerants on boats. Aquaculture studies included feed and energy use up to the point of harvest, except for the salmon study which included feed only (Pelletier and Tyedmers, 2007). For transport of imports by boat to Australia we included fuel use for the journey and the refrigerated container for frozen products. Harbour activities have not been included. For airfreight we model fuel use for the journey.

Sensitivity analysis is also performed on the aspects considered to have the greatest impact on overall results: feed conversion ratio (FCR) and catch per unit effort (CPUE). For aquaculture species we model the effect of lowering or raising the FCR on the CF, assuming other factors remain the same including feed composition, and energy use and emissions associated with feed production. For wild-capture species we model the impact on results of changes in fuel use over time as a result of changing CPUE. We make the following assumptions: (1) that the catch rate scales effort and therefore fuel use; (2) that all emissions are perfectly variable with catch rates and there are no fixed emissions, i.e. the fleet will rescale with catch rate, for example, if catch rate doubles then fuel emissions halve, because trips, bait, and gear required halve; (3) that the fleet is homogenous so when the fleet rescales with catch rate an average vessel enters or leaves the fishery and the composition of the fleet, in terms of efficiency of individual vessels, stays the same.

3. Results

3.1. Carbon footprint of seafood in Australia

The capture or farm stage was typically the major source of carbon emissions for frozen seafood transported by sea. Carbon emissions from seafreight were less than 1 kg CO_2e kg^{-1} seafood (Fig. 1). For prawns, these emissions accounted for 4% of the CF for transport from China and the Philippines to Australia (Fig. 1a). Trap-caught *Homarus americanus* landed in the USA and shipped to Australia had a smaller CF at wholesale in Sydney than did the Australian southern rock lobster both at landing in Tasmania and at wholesale in Sydney (Fig. 1b). It is notable that the CF increased by over 400% or 18 kg CO_2e kg^{-1} when *H. americanus* was flown (Boston–Los Angeles–Sydney, main flight path), instead of shipped (Boston–Middle East–Sydney, main sea route) to Australia from the USA. Only small amounts of lobster are currently flown to Australia from the USA, all of which are frozen.

Emissions from seafreight accounted for less than 10% of total emissions for catfish from Vietnam, canned salmon from the USA, and hake from Spain, while they accounted for over 60% for sardines from Portugal (Table 1). For fish species with low emissions at the production stage, a modest increase in total carbon emissions from seafreight resulted in substantial percentage increases in the CF. The addition of seafreight to 1 kg of sardines from Portugal, for example, resulted in a 157% increase of the CF despite only increasing emissions per kilogram sardine by approximately half a kilogram CO_2e.

Transport of frozen salmon from the USA to Australia resulted in emissions of 0.7 kg CO_2e kg^{-1}, which accounted for 25% of the CF (Fig. 1c). The transport stage of canned salmon, which does not require refrigeration, accounted for 0.3 kg CO_2e kg^{-1}, while the canning process was responsible for 6.6 kg CO_2e kg^{-1} or 73% of carbon emissions (Fig. 1c). The farming of salmon accounted for 32% of the CF for canned salmon and 82% for frozen salmon. For airfreighted salmon from the USA, carbon emissions increased by 18 kg CO_2e kg^{-1} relative to seafreight (Fig. 1c). Airfreight accounted for 57% of the CF for catfish from Vietnam and 1 kg of airfreighted catfish had a CF 12 kg CO_2e larger than if sent by sea. Seafreight of catfish, in contrast, accounted for only 2% of carbon emissions and resulted in 0.2 kg CO_2e kg^{-1} (Table 1).

3.2. Comparison of carbon footprint at landing or harvest by species

Carbon emissions varied between different species of fished and farmed seafood. Wild-caught *Penaeus esculentus*, an endemic Australian prawn, had the highest CF of all the seafood examined in this study (see supplementary material for full list of species), accounting for 32 kg CO_2e kg^{-1} (Farmery et al., 2015) (Fig. 2a). Farmed *P. monodon* prawns had lower emissions at 5.1 kg CO_2e kg^{-1} (Baruthio et al., 2008), similar to that of farmed *Litopenaeus vannamei*, 3.1 kg CO_2e kg^{-1} (Cao et al., 2011). Emissions related to wild-caught banana prawns (*F. merguiensis*) were similar to that of farmed prawn species, at 4.2 kg CO_2e kg^{-1} (Farmery et al., 2015). *J. edwardsii* lobsters from Australia, had

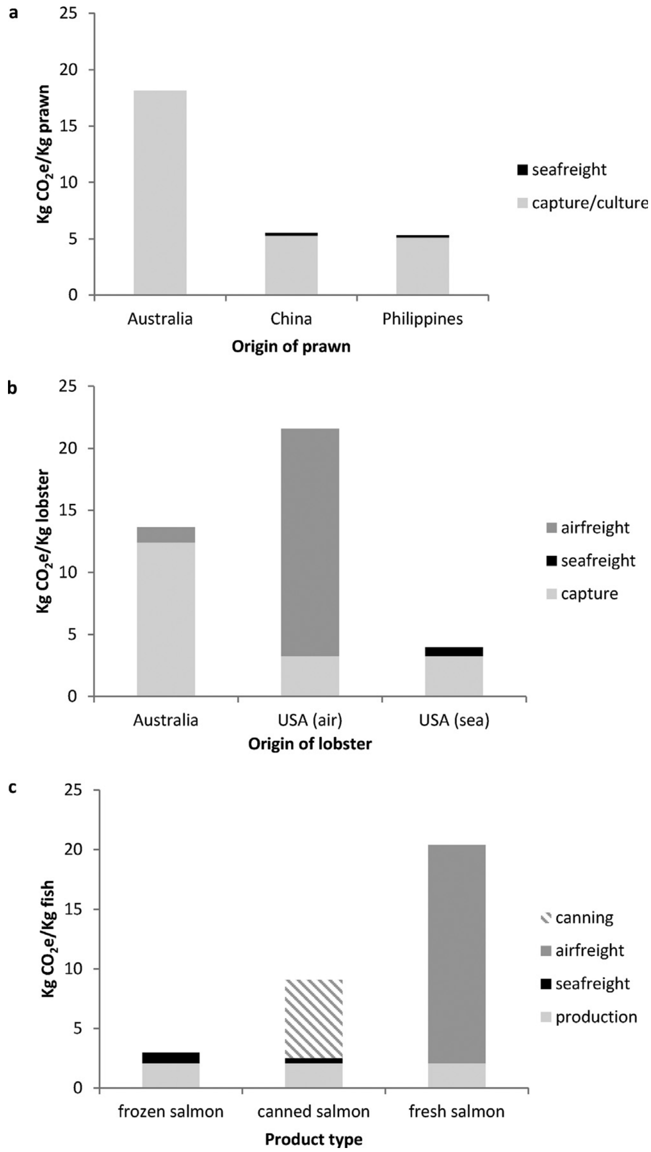


Fig. 1. Carbon footprint of 1 kg whole seafood with supply chain stages*: (a) whole frozen prawn with refrigerated seafreight to Australia; (b) whole frozen lobster: *H americanus* with sea- and airfreight to Sydney, and live *J. edwardsii*; (c) whole Atlantic salmon with canning and transport – frozen salmon includes refrigerated transport, fresh salmon does not include refrigeration. *See Fig. 2 for sensitivity of results by species.

higher carbon emissions, 12.3 kg CO_2e kg^{-1} (Farmery et al., 2014), than *H. americanus*, 4.4 kg CO_2e kg^{-1} (average of USA and Canada) (Boyd, 2008; Driscoll, 2008) (Fig. 2b).

Catfish (*Pangasianodon hypophthalmus*) had the highest CF of all fish species at 9 kg CO_2e kg^{-1} (Bosma et al., 2011) (Fig. 2c). Hake (*Merluccius merluccius*) had a lower footprint than catfish, 5.3 kg CO_2e kg^{-1} (Iribarren et al., 2010) but a higher footprint than frozen salmon (*Salmo salar*) and flathead (*N. richardsoni*) which both had emissions around 3 kg CO_2e kg^{-1} (Farmery et al., 2015; Pelletier and Tyedmers, 2007). Horse mackerel (*Trachurus trachurus*) had emissions of 2.4 kg CO_2e kg^{-1} (Iribarren et al., 2010) while sardines (*Sardina pilchardus*) and Australian salmon (*A. trutta*) had a CF of 1 kg CO_2e kg^{-1} or less (Almeida et al., 2014; Farmery, 2015; Vazquez-Rowe et al., 2014), the smallest of all seafood examined (see Section 3.3 for sensitivity of results).

Table 1
Carbon emissions for different fish products at production, processing and transport.[f]

Fish	Origin	Production $CO_2e\ kg^{-1}$	Sea freight $CO_2e\ kg^{-1}$	Air freight $CO_2e\ kg^{-1}$	Canning $CO_2e\ kg^{-1}$	Total $CO_2e\ kg^{-1}$	Production % total	Transport % total	Canning % total
Catfish (*Pangasianodon hypophthalmus*)[a]	Vietnam	8.9	0.2			**9.1**	98	2	
Catfish (*P. hypophthalmus*)[a]	Vietnam	8.9		7.7		**17**	54	46	
Hake (*Merluccius merluccius*)[b]	Spain	4.8	0.53			**5.3**	90	10	
Flathead (*Neoplatycephalus richardsoni*)[c]	Australia	2.4				**2.4**	100		
Frozen salmon (*Salmo salar*)[d]	USA	2.1	0.7			**2.8**	75	25	
Canned salmon (*S. salar*)[d]	USA	2.1	0.3		6.6	**9**	24	3	73
Fresh salmon (*S. salar*)[d]	USA	2.1		18.3		**20**	10	90	
Horse mackerel (*Trachurus trachurus*)[b]	Spain	1.85	0.53			**2.6**	72	28	
Australian Salmon (*Arripis trutta*)[c]	Australia	0.97				**1**	100		
Sardine (*Sardina pilchardus*)[b]	Spain	0.74	0.53			**1.3**	58	42	
Sardine (*S. pilchardus*)[e]	Portugal	0.36	0.56			**0.9**	38.9	61	

[a] Bosma et al. (2011).
[b] Iribarren et al. (2010).
[c] Farmery (2015).
[d] Pelletier and Tyedmers (2007).
[e] Almeida et al. (2014).
[f] Sensitivity of results by species presented in Fig. 2.

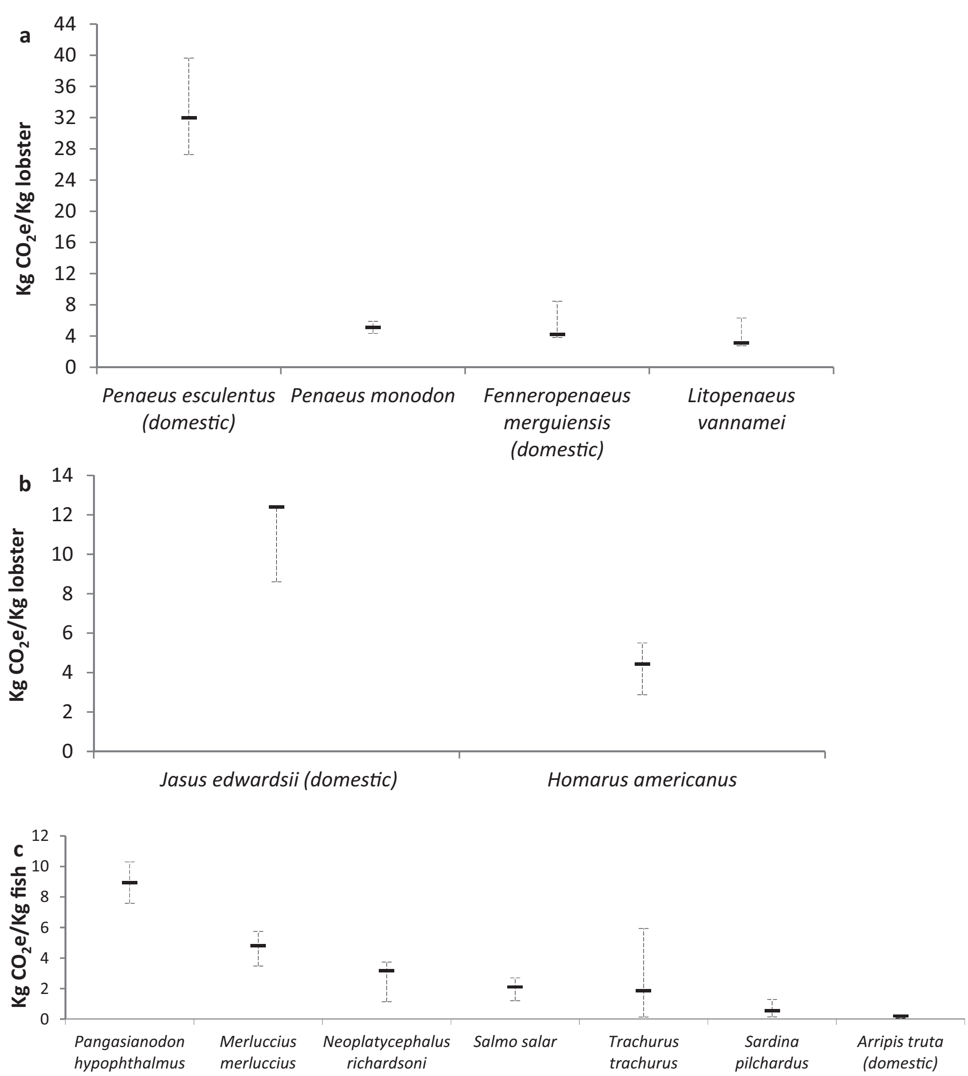


Fig. 2. Carbon footprint of 1 kg whole prawn, whole lobster and whole fish at landing/farm gate. Error bars represent range of reported results based on variation in catch per unit effort over time for wild-capture and different feed conversion ratios for aquaculture.

3.3. Comparison of carbon footprint at landing and harvest by production method

Three different types of prawn aquaculture had lower CF than the Australian trawl caught prawns: polyculture (Baruthio et al., 2008), intensive, and semi-intensive (Cao et al., 2011) (Fig. 3a). Emissions from the two prawn trawl fisheries were averaged at 18 kg CO_2e kg^{-1} (Farmery et al., 2015), however, the CF of trawling for banana prawns was similar to that of aquaculture prawns. All lobsters were trap caught therefore no comparison between methods was made.

There was substantial variation between fishing methods for finfish reported in the literature, although studies on different methods were not available for all species. Flathead caught by otter trawl in Australia had a footprint more than double those caught by Danish seine, 3.5 kg CO_2e kg^{-1} compared with 1.3 kg CO_2e kg^{-1} (Farmery, 2015). The CF of pond aquaculture catfish (Bosma et al., 2011) was larger than net pen aquaculture salmon (Pelletier and Tyedmers, 2007). Marine aquaculture and passive gear, such as purse and Danish seine, had the lowest CF for gear and production types (Fig. 3b).

3.4. Sensitivity analysis

3.4.1. Feed conversion ratio (FCR)

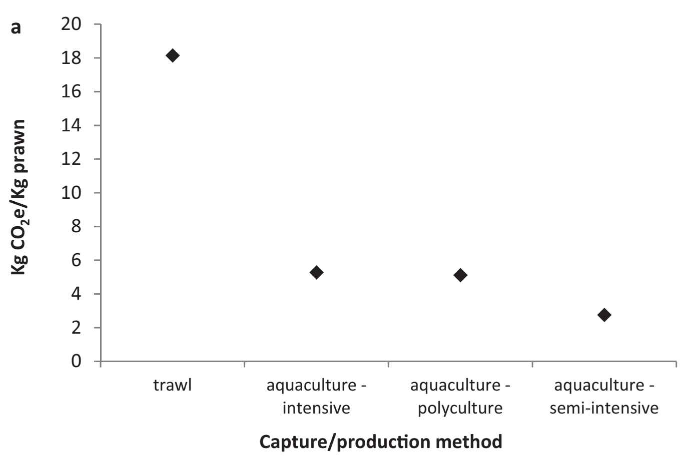
Results for farmed *L. vannamei* varied from 2.75 to 6.3 kg CO_2e kg^{-1} (Fig. 2a) based on the FCR range for semi-intensive and intensive aquaculture presented by Cao et al. (2011). The CF range for *Penaeus monodon* varied from 4.34 to 5.88 kg CO_2e kg^{-1} based on ±15% to account for potential changes in fuel use for collecting snails for feed (Baruthio et al., 2008) (see Table S1.7). The ranges presented remain comparable to wild-capture *F. merguiensis* and substantially lower than *P. esculentus*. The standard deviation of FCR for *P. hypophthalmus* (Bosma et al., 2011) was used to determine a range of carbon emissions. When ranges are considered for all fish species, the production of *P. hypophthalmus* remains the most carbon intensive per kilogram. Pelletier et al. (2009) provide a FCR range for salmon which we use to calculate the CF range of 1.78–2.42 kg CO_2e kg^{-1} for *Salmo salmar*.

3.4.2. Catch per unit effort (CPUE)

CPUE for tiger prawns at the time of the study was 0.15 t/day, which was also the mean CPUE from 2004 to 2013. Results indicate that when CPUE is higher than 0.15, *P. esculentus* remains more carbon intensive per kilogram than other prawn species. CPUE for *F. merguiensis*, 1.96 t/day, was higher than the mean for 2004–2013 of 1.5. The CF of 4.2 kg CO_2e kg^{-1} used here is therefore slightly lower than the average for the past decade although the CF for *F. merguiensis* remained similar to that of aquaculture prawns.

The CPUE for *J. edwardsii* is at an 11-year low and the CF presented here is higher than it may have been in previous years. The CF of *J. edwardsii* with high CPUE remains larger than *H. americanus*, however, the footprints are more comparable when CPUE for *H. americanus* is low (Fig. 2b).

When the CF range was examined for all fish species, *M. merluccius*, *N. richardsoni*, *T. trachurus* and *S. salmar* were more carbon intensive per kilogram than small pelagics and less than *Pangasius*. CPUE for *T. trachurus* varied from 0.2 to 7.8 t/fishing trip between 1995 and 2012 resulting in a large range in the CF of 1.85 kg CO_2e kg^{-1} presented here.



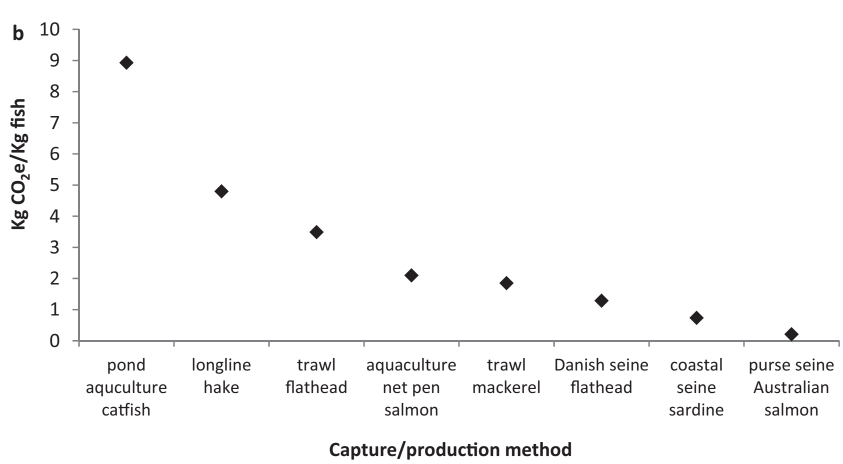


Fig. 3. Carbon footprint of 1 kg whole prawn and whole fish with different capture and production methods. See Fig. 2 for sensitivity of results by species.

3.4.3. System boundaries

The CF varied between 3.8–5 CO_2e kg^{-1} for wild-caught banana prawn and 29–39 CO_2e kg^{-1} for wild-caught tiger prawn depending on the assumptions made about the inclusion of refrigerants and fuel use for freezing (Table S1.9). The range of results averaged across the two wild-capture fisheries was 16–22 CO_2e kg^{-1}. The inclusion of on-farm activities for net-pen salmon resulted in a 6% increase in carbon emissions and the CF at production rose by 1.3 kg CO_2e kg^{-1} (Table S1.10), which was higher than the range presented in Fig. 2c.

4. Discussion

4.1. Carbon footprint of seafood

Our results show that seafood imported into Australia does not necessarily have a higher carbon footprint (CF) than domestically produced seafood, despite the increased distance between production and consumption. It reiterates previous research that food miles, or distance travelled, are not the most accurate measure of impact (Coley et al., 2013; Edwards-Jones et al., 2008; Garside et al., 2008; Hogan and Thorpe, 2009; Weber and Matthews, 2008; Wynen and Vanzetti, 2008) and that production and transportation mode are more important considerations than distance (Avetisyan et al., 2014). Imported products can have comparable or in some cases smaller CF than domestic products. Seafood produced on the other side of the globe, frozen and shipped, may be the most energy efficient (Tlusty and Lagueux, 2009), an important consideration for sustainable food policy.

For example, we found that the CF of USA lobster (*H. americanus*) on arrival in Sydney was lower than that of the locally produced Tasmanian southern rock lobster (at landing and after airfreight to wholesale) despite travelling approximately 29,000 km from East coast of the USA by refrigerated container. In contrast, New Zealand is a major supplier of fresh fish to Australia, and although the two countries are neighbours, much of this food is airfreighted and therefore has a higher footprint than some frozen and processed fish transported from further away by sea.

This concept can be explored through the example of carbon emissions associated with plates of seafood consumed in Australia, consisting of 150 g of fish, prawn and lobster meat (Fig. 4). The footprint of a plate of 150 g of Australian banana prawns, southern rock lobster and Australian salmon (50 g each of edible meat) is 1.5–2.5 kg CO_2e. The footprint of a similar plate of imported seafood including 150 g of white-leg shrimp, catfish and American lobster is comparable at 1.5–2 kg CO_2e. In another example, the CF of a plate made up of Australian wild-capture tiger prawns, southern rock lobster and flathead is between 4 and 6 kg CO_2e, while the CF of a plate made up of imported tiger prawns, American lobster and sardines is only 1 kg CO_2e. The seafood plates compared in these examples are not perfect substitutes, and a comprehensive comparison should account for other sustainability issues, cost, consumer preference, and include other popular species eaten in Australia as well as variations in CF over time. These examples are used here to demonstrate that for consumers and policymakers concerned about carbon footprints of food, imported seafood can be competitive with domestically produced goods.

Local food production is associated with many positive values (Schnell, 2013), however, the use of food miles as a sustainability metric ignores other supply chain stages and environmental considerations, potentially overshadowing more relevant indicators that are important for balanced debate on food sustainability (Avetisyan et al., 2014). Our finding that the production stage (capture or culture), not transport, is typically the major contributor to the CF of seafood products is consistent with the LCA literature (Cao et al., 2011; Hospido and Tyedmers, 2005; Pelletier and Tyedmers, 2010; Thrane, 2006; Vázquez-Rowe et al., 2013; Ziegler and Valentinsson, 2008). For wild-capture fisheries, the size of the CF is a reflection of the fuel efficiency of fishing boats, which is determined by the species targeted and the type of gear used (Schau et al., 2009; Tyedmers, 2001; Tyedmers and Parker, 2012) as well as fisher behaviour and management regime (Farmery et al., 2014; Vázquez-Rowe and Tyedmers, 2013). Management in particular can influence biomass and effort in fisheries which can in turn change fuel use (Parker et al., 2015; Ziegler and Hornborg, 2014).

CF of aquaculture species can also vary with farming system. Intensive aquaculture of white-leg shrimp in China, for example, had a higher CF than semi-intensive systems due to higher on-farm energy and feed use (Cao et al., 2011). The degree of intensification, however, may not be as important as other factors such as system efficiency for distinguishing the impacts of aquaculture systems (Aubin et al., 2015). Feed use is a pivotal driver of environmental performance (Pelletier et al., 2009) as seen through large-scale

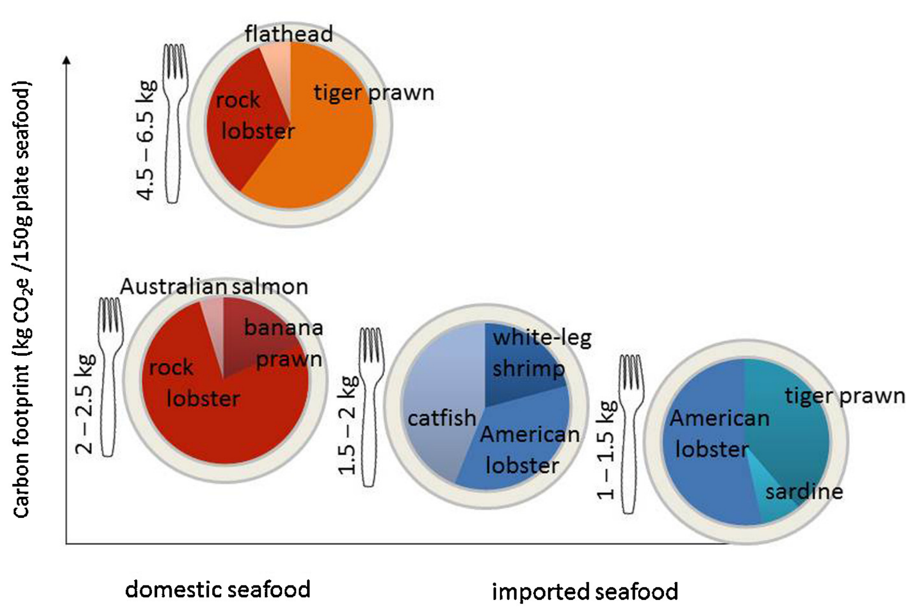


Fig. 4. Carbon footprints of plates made up of 150 g edible seafood from different domestic and imported sources.

production of tilapia and carp where efficient feeding practices resulted in lower carbon emissions per kilogram than small-scale farming (Mungkung et al., 2013). When production methods are compared, the literature supports our finding that marine-based aquaculture systems have a comparatively low CF, a function of being less energy-intensive than land-based systems (Ayer and Tyedmers, 2009), while pond-based aquaculture can have a higher CF as a result of aeration required to maintain water quality (Pelletier and Tyedmers, 2010).

Long-haul airfreighted products have higher CF than non-airfreighted products (Andersen, 2002; Farmery et al., 2014; Winther et al., 2009). Most seafood exported to Australia is sent by sea (www.abs.gov.au) and airfreight is generally required for highly perishable products, where no processing or storage infrastructure exists. Globally, 90% of trade in fish and fishery products consists of processed products (FAO, 2012) which negates the need for airfreight and refrigeration, although a trade-off exists where the processing stage contributes to the life cycle impacts of a product. The canning process adopted from Almeida et al. (2015) in this study was the main source of carbon emissions of canned salmon. The footprint of canned products was greater than frozen products predominantly due to the use of tin for the cans. Canning also represented the largest contribution to the CF of tuna (Hospido et al., 2006) and sardine (Vazquez-Rowe et al., 2014) supply chains. The disposal of packaging materials used to ship frozen catfish fillets has also been identified as an area for improvement (Nhu Thuy et al., 2015). Processing, usually nonetheless, prolongs product shelf-life which is an important consideration given that food wastage from storage, handling, transport and final consumption can be as high as 50% for seafood in countries in North America and Oceania (Gustavsson et al., 2011).

Carbon emissions are not a traditional measure of seafood sustainability yet the impacts from climate change, including ocean acidification and rising water temperatures (IPCC, 2014), may present a greater threat than the localised production impacts currently informing sustainability assessments. LCA may not capture all of the sustainability issues posed by a globalised, highly complex food system (Garnett, 2009) or some unique fishery impacts (Curran et al., 2010; Pelletier et al., 2007), however, opportunity exists to combine the assessment of impacts considered under current measures of sustainability with impacts such as carbon emissions. This combination would provide a more holistic understanding of seafood sustainability as well as highlighting the compatibility or trade-offs between different sustainability goals.

4.2. Current seafood sustainability assessment of wild-capture seafood

Demersal trawling can be responsible for ecosystem impacts (Lack, 2010) and some of the highest CF of all fishing methods. Opportunity therefore exists to improve the localised ecological impacts of trawling as well as broader environmental impacts through improved fisheries management (Driscoll and Tyedmers, 2010; Farmery et al., 2014; Ziegler and Hornborg, 2014). Fuel use efficiency in the Australian Northern Prawn fishery, a global model for many aspects of fisheries management (Gillett, 2008), has been improving (Pascoe et al., 2012) although reducing carbon emissions has not been a management goal. Prawn trawl fisheries in Senegal, in contrast, are potentially less well managed, given that Senegal was ranked alongside the worst performing countries in an assessment of compliance with the FAO Code of Conduct for Responsible Fishing (Pitcher et al., 2008) and the CF of trawl caught Senegalese pink shrimp, while not directly comparable to the Australian example, was reportedly high (Ziegler et al., 2011). Management, and its influence on carbon emissions, may be a more important consideration for seafood sustainability than the distance a product has travelled.

4.3. Current seafood sustainability assessment of aquaculture

While some farmed seafood can have a lower CF than wild-capture species, there is a range of other environmental impacts associated with aquaculture. Mangrove loss, pollution of agricultural land and water, and impacts on wild fish stocks from wild seed stock collection and feed have all been documented (see for example Ahmed et al., 2010; Diana, 2009; Jonell and Henriksson, 2015; Naylor et al., 2000; Páez-Osuna, 2001). The use of fishmeal has also been identifying as the overall largest single contributor to the CF of the Asian aquaculture sector (Henriksson et al., 2014). However, the majority of seeds are now artificially produced and there have been recent advances in replacement of fishery products in shrimp diets (Glencross et al., 2014).

Several third-party aquaculture assessments have emerged such as the Global Aquaculture Alliance Best Aquaculture Practice certification programme. However, the success of the programme in reducing environmental impacts is unknown (Tlusty and Tausig, 2014). Energy consumption and carbon emissions of farms are included in the Aquaculture Stewardship Council (ASC) standards and a Responsible Feed Standard is currently being developed (http://asc-aqua.org). This inclusion of the life cycle perspective demonstrates how broader environmental considerations are beginning to be incorporated into seafood sustainability.

Whether seafood products are produced near or far from where they are consumed should not be the main consideration for assessment of their relative sustainability (Tlusty and Lagueux, 2009). Instead there needs to be a focus on the whole system – covering production, distribution and consumption. Policy decisions designed to negate the environmental costs of food production through reduced meat consumption, while nourishing a burgeoning populace (Eshel et al., 2014; Garnett, 2009), may unintentionally lead to the greater promotion of seafood to meet recommended protein intakes. Policy makers will need to examine existing sustainability criteria, as well as broader impacts associated with species type, production method and distribution mode, when considering seafood and sustainability within food policy.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.envsci.2015.06.007.

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