Advances in Water Resources 110 (2017) 505–514

Contents lists available at ScienceDirect

Advances in Water Resources

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

The 'seafood gap' in the food-water nexus literature—issues surrounding freshwater use in seafood production chains

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ARTICLE INFO

Article history:
Received 13 September 2016
Revised 3 February 2017
Accepted 14 March 2017
Available online 16 May 2017

Keywords:
Aquaculture
Fisheries
Food-water nexus
Freshwater use assessment
Impact assessment methodology
Water footprint

ABSTRACT

Freshwater use for food production is projected to increase substantially in the coming decades with population growth, changing demographics, and shifting diets. Ensuring joint food-water security has prompted efforts to quantify freshwater use for different food products and production methods. However, few analyses quantify freshwater use for seafood production, and those that do use inconsistent water accounting. This inhibits water use comparisons among seafood products or between seafood and agricultural/livestock products. This 'seafood gap' in the food-water nexus literature will become increasingly problematic as seafood consumption is growing globally and aquaculture is one of the fastest growing animal food sectors in the world. Therefore, the present study 1) reviews freshwater use concepts as they relate to seafood production; 2) provides three cases to highlight the particular water use concerns for aquaculture, and; 3) outlines future directions to integrate seafood into the broader food-water nexus discussion. By revisiting water use concepts through a focus on seafood production systems, we highlight the key water use processes that should be considered for seafood production and offer a fresh perspective on the analysis of freshwater use in food systems more broadly.

1. Introduction

The freshwater resource requirements of food production inextricably link food and water security. As a result, meeting the basic food and water needs of a growing human population with changing diets is a central concern of resource analysts (UNEP, 2012) and connects particularly to two of the Sustainable Development Goals: to end hunger and ensure access to water (UN, 2015). This global challenge has prompted a growing literature on the food-water nexus. This literature includes analyses of opportunities to improve the joint food-water security, which often involves quantifications of freshwater use for food production. However, the interaction of food production and water resources is complicated by the fact that water uses vary in how they influence the quality and quantity of water downstream (Table 1).

It has been estimated that agriculture and livestock production currently account for 60% of global freshwater withdrawals and more than 90% of consumptive use (World Bank, 2014; Moore et al., 2015). Water use estimates for livestock have been particularly high and global livestock consumption is projected to increase with growing GDP and populations. Correspondingly, water use for livestock production is anticipated to increase 65% by 2050 under a business-as-usual scenario (Davis et al., 2016). In order to limit use, several studies promote improved production efficiency, reduced food waste, and alternative diets, such as vegetarian or pescetarian diets, to reduce the environmental burden of food production while satisfying the growing global demand (Foley et al., 2011; Tilman and Clark, 2014; Davis et al., 2016; Gephart et al., 2016).

Regardless of whether these recommendations lead individuals to adopt diets that include more freshwater and marine fish and other aquatic animals (henceforth 'seafood'), seafood demand is projected to increase with growing populations and demographic shifts (Delgado et al., 2003; Tilman et al., 2011; FAO, 2016a). Since

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http://dx.doi.org/10.1016/j.advwatres.2017.03.025

Table 1
Glossary defining types of freshwater use and impacts. From: Bayart et al. (2010), Hoekstra et al. (2011) and Bayart et al. (2014).

Terminology	Definition
Freshwater use	Generic term for all types of human freshwater resource use
In-stream freshwater use	Use of fresh water in situ (e.g., navigational transport on a river)
Off-stream freshwater use	Use of fresh water that requires human removal from a natural body of water or groundwater aquifer (e.g., pumping or diversion of water for municipal, agricultural, or industrial purposes)
Freshwater degradative use	Withdrawal of fresh water and discharge into the same watershed after the quality of the water has been altered (includes both quality deterioration and improvement)
Freshwater consumptive use	Use where fresh water is not returned to the original watershed because of evaporation, product integration, or discharge into different watersheds or the sea*
Water quality impact	Changes in quality of water entering versus exiting a process
Competition for freshwater resources	Temporary reduction of freshwater resources available for alternative uses
Freshwater depletion	Net reduction in the availability of fresh water in a watershed and/or fossil groundwater stock. Depletion occurs when freshwater consumptive use exceeds the renewability rate of the resource over a significant time period
Blue water	Surface and groundwater consumed
Green water	Rainwater consumed
Grey water	Fresh water needed to dilute pollutants to background concentrations or meet existing water quality standards

*Hoekstra et al. (2011) also includes water which does not return to the watershed in the same period.

1961 seafood consumption has grown at an annual rate of 3.6%, or twice the rate of human population growth (WHO, 2014), and in 2013 3.1 billion people obtained at least 20% of their animal-derived protein from seafood, as well as essential omega-3 fatty acids and micronutrients (Beveridge et al., 2013; Tacon and Metian, 2013; FAO, 2016a). During this period there have also been dramatic changes in the seafood industry. Aquaculture production now comprises half of global seafood production, target and cultivated species have shifted, and production and trade have become increasingly globalized (Deutsch et al., 2011; Gephart and Pace, 2015; FAO, 2016a).

With the growth and change of the aquaculture sector, its dependence on, and conflicts related to, freshwater resources have come in focus. For example, recent drought reports in Thailand indicate that 200 fish farms were forced to close due to lack of available water resources, while flooding at other times has destroyed fish ponds (Suvansombut, 2015). Meanwhile, groundwater extraction for fish farms in China's Yellow River Delta is causing subsidence at rates as high as a quarter meter per year (Higgins et al., 2013). There have also been disputes over freshwater access, with Egyptian tilapia farmers denied use of irrigation water, pollution from Chinese farms degrading fresh and coastal waters, and accusations of shrimp farms causing saltwater intrusion in Bangladesh and Thailand (Szuster and Flaherty 2002; Cao et al., 2007; Azad et al., 2009; Eltholth et al., 2015).

Inland capture fisheries have also come in conflict with other water uses. Damming rivers to hold water for irrigation or hydropower adversely impacts freshwater capture fisheries and the proposed dams in the Lower Mekong, Congo, and Amazon have been projected to substantially reduce local fish catches (Orr et al., 2012; Winemiller et al., 2016), although there is a potential trade-off since reservoirs can provide an environment for aquaculture. At a broader food system level, there are high water costs associated with replacing these capture fisheries, which require few freshwater inputs, with agricultural production (Orr et al., 2012). Gephart et al. (2014) found that replacing marine capture seafood with terrestrial foods would increase the global water footprint by 4.6%, with larger increases of 20–50% in Asia, Oceania, and coastal African nations, although the agriculture production (and therefore the impacts on water resources) may occur in a geographically-distant location. Such high replacement water costs would be most problematic for countries with few domestic water resources and limited ability to import substitute foods (Gephart et al., 2014).

Despite future outlooks, freshwater use in aquaculture and fisheries has received little attention in the scientific literature, with most global water use analyses focusing on agriculture, industry, and domestic water use. We contend that in part this is because no consistent methodology exists for evaluating freshwater use in aquaculture, and key data necessary for estimating this water use are lacking. As seafood consumption continues to increase globally, the 'seafood gap' in the food-water nexus literature will become increasingly problematic. This inhibits analyses of potential synergies and trade-offs between seafood and terrestrial foods in terms of water use, future food-water scenarios, informed decision-making, and in policy considerations. Specifically, analyses of trade-offs in diets and of diet scenarios (e.g. Tilman and Clark, 2014; Gephart et al., 2016; Davis et al., 2016) and proposed environmental labeling and certification schemes require detailed inventory data and meaningful methodology across the full range of products (Jonell et al., 2013; Leach et al., 2016). Further, policy proposals that aim to regulate freshwater consumption, such as a water market, would need to evaluate the potential impacts by sector. Such an analysis requires a uniform method for freshwater accounting that includes seafood production. Without this information, the potential impacts on (and from) the growing aquaculture sector may remain unaccounted for or insufficient water may be reserved for environmental flows that support capture fisheries.

Prompted by these needs, the present study 1) reviews water use concepts as they relate to seafood production; 2) provides three cases to highlight the particular water use concerns for aquaculture, and; 3) outlines future directions for seafood to be incorporated into the broader food-water nexus discussion. In this study, we emphasize there is a range of existing approaches to quantify water use and the appropriate method depends on the researcher's specific question. When making relative comparisons across products, it is crucial to include the most important water uses for each product under consideration. While our primary focus is on water quantity issues, we discuss quality issues in Section 5. By evaluating the primary freshwater uses in seafood production, we provide guidance on the necessary considerations for comparative analyses which include seafood and offer a fresh perspective on the analysis of freshwater use in food systems more broadly.

2. Freshwater use metrics background

Several methodologies have been suggested to quantify freshwater use (SI Table 1). The first suggestion to quantify water use over production chains and scale it to a unit of reference was made by Guinée et al. (1993) as a potential impact category in the life cycle assessment (LCA) framework. While water use could be scaled to multiple possible units of reference (e.g. product or whole system), we focus our discussion here on product-level analyses. Scaling to the product level enables comparison of product-level water use, but necessitates allocation decisions, which must be clearly stated and well justified. Independent of the water use impact category in LCA, the concept of 'virtual water' flows emerged as a construct to conceptualize freshwater consumption attributable to products, especially food commodities (Allan, 1996). Hoekstra and Hung (2002) used the virtual water concept to make the first global evaluation of virtual water flows embedded in crop trade based upon evapotranspiration. In parallel, the LCA community emphasized classifying the origin and receiving body of different types of freshwater uses. Starting from classifications based upon the regeneration potential of different sources, including deposits (fossil groundwater), funds (aquifers and lakes) and flows (rivers and streams) (Koehler, 2008); detailed methods were developed to account for water scarcity, ecosystem quality, human consequences and water quality (Pfister et al., 2009; Bayart et al., 2014). This approach, however, requires information on the location of the water withdrawal, and the amount (and dynamics) of water locally or regionally available, in addition to the amount of water used (Guinée et al., 1993). In response, the simplified blue, green, and grey (Table 1) water footprint concept was developed by the Water Footprint Network (WFN, waterfootprint.org) and is now favored by many as it is easier to operate (Hoekstra et al., 2011).

Water use quantification methods can be compared based on the types of flows included in consumptive use, the system boundaries, the inclusion of scarcity metrics, and the assumptions and calculation choices. Freshwater consumptive use includes water that is not returned to the original watershed because of evaporation, product integration, or discharge into different watersheds or the sea (Table 1). Hoekstra et al. (2011) uses a similar definition, but also includes water that does not return to the watershed in the same period. As a result, the focus in agriculture systems tends to be on evaporative losses. For example, the WFN uses a water balance model for agriculture that considers daily soil water use, crop water requirements, crop water use, and yields to determine water footprints (Mekonnen and Hoekstra, 2011). The WFN livestock commodity water footprints then include both the indirect water footprint of agricultural crops used as feed and the direct water use in form of drinking and service water (Mekonnen and Hoekstra, 2012a). LCA methods also tend to emphasize freshwater consumptive use related to evaporative losses, but with less emphasis on green water since evapotranspiration from soil water occurs also with natural vegetation (and use then represents a change in evapotranspiration).

Where the system boundaries are set determines which processes are included in an analysis and which are not. It is important for the system boundaries to align with the researcher's question and include the major water using processes for all products being considered. Methods can differ in the number of product life stages being considered (e.g. cradle to gate versus cradle to grave), and in the processes which are assumed to be insignificant. The model used to produce the WFN product database standardizes the system boundaries across many products, while the software and inventory databases which support many LCA studies allow for more processes to be included within the system boundary, such as industrial processes.

A persisting methodological issue for both LCA and the WFN approaches is relating use to scarcity. Although water resources are renewable, there is a fixed quantity that can be allocated at a given point in time. As a result, competition among different users (e.g. food production, industry, ecosystems) becomes more problematic in situations of scarcity. The WFN approach has been criticized for lacking water-scarcity weighting and therefore being unable to reflect the potential local environmental impact of an activity's water consumption (Pfister and Hellweg, 2009). This view mainly originates from the LCA community and has created a discussion that is still active (Hoekstra et al., 2009; Hoekstra, 2016). Incorporating scarcity can also be difficult due to the lack of data on the specific locations of most food system unit processes in globalized markets.

Independent of the preferred approach, some pivotal choices remain and need to be well-argued and defined, such as how to allocate water consumption among several products originating from a common process (e.g. fish fillets and fish heads from fish processing). In the field of LCA, this is a topic of discussion that has dragged on for three decades without reaching a consensus (Henriksson et al., 2012a), while the WFN tends to rely on allocation by monetary value. Indifferent to allocation method, the solution should be well justified, in-line with the main methodological principles and consistently applied to all allocation scenarios (Guinée et al., 2004; JRC, 2010). Moreover, consistent and clear freshwater use nomenclature and methodology are required to enable comparisons among food commodities.

3. Freshwater use in aquaculture

Previous studies have evaluated a range of aquaculture production environmental impacts (e.g., Naylor et al., 2000; Costa-Pierce, 2010; Samuel-Fitwi et al., 2012). Several studies have also used LCA to analyze environmental performance of aquaculture, including its water use throughout whole production cycle (for a review, see Henriksson et al., 2012a), the WFN approach to estimate the water footprint associated with aquafeeds (Troell et al., 2014a; Pahlow et al., 2015; Gephart et al., 2016), and a hydrology-based accounting for inland aquaculture (Boyd and Gross 2000; Boyd 2005). Despite their methodological differences, these studies demonstrate that freshwater use can be large for some aquaculture systems, but that freshwater consumption varies greatly depending on production system, location, and species being produced (Verdegem and Bosma 2009; Verdegem et al., 2006; Phillips et al., 1991).

From a freshwater use perspective, aquaculture production systems can be divided into recirculating, semi-closed, and open water systems. Recirculating systems are systems that have no water seepage and mainly lose water through evaporation. Semi-closed systems (e.g. ponds) consist of manmade water embankments that rely on natural water sources, either fresh or brackish (Tidwell, 2012). The main direct water losses from ponds are evaporation, seepage, and water exchange (Fig. 1) and can be represented by standard hydrologic equations (Yoo and Boyd, 1994). The local environment of the pond contributes to both the evaporation and seepage rates and contributes to the variability in water use in semi-closed systems. Open water systems refer to systems that utilize existing water bodies for production, including stocked natural bodies of water, pens or cages, and shellfish growing racks. In these systems, the evaporation (or flow through water volume) does not represent freshwater consumption since the open water surface area existed prior to the farming practices and the practice itself does not change the evaporation rate (Fig. 1). Although water passing through the cages is not consumed, its quality may be altered, representing a degradative use (discussed further in Section 5). The consumptive use in open water systems is instead generally dominated by indirect use (e.g. Mungkung et al., 2013).

Indirect water use for aquaculture depends on the system boundaries considered, but generally includes all or some of the water use for aquafeed production, processing of farmed products, and energy generation (Fig. 1). These factors depend on the species, the final product, and the system. This is because there is high variability in the dependence on and composition of commercial aquafeeds for different species. For example, 95–100% of salmon, trout, and shrimps rely on feeds, compared to moderate levels of feed use for carps and other omnivorous fishes and none for farmed bivalves (Tacon et al., 2011; Troell et al., 2014b). The composition of feeds and economic feed conversion ratios (eFCRs) also vary by species. Together these result in seafood water foot-

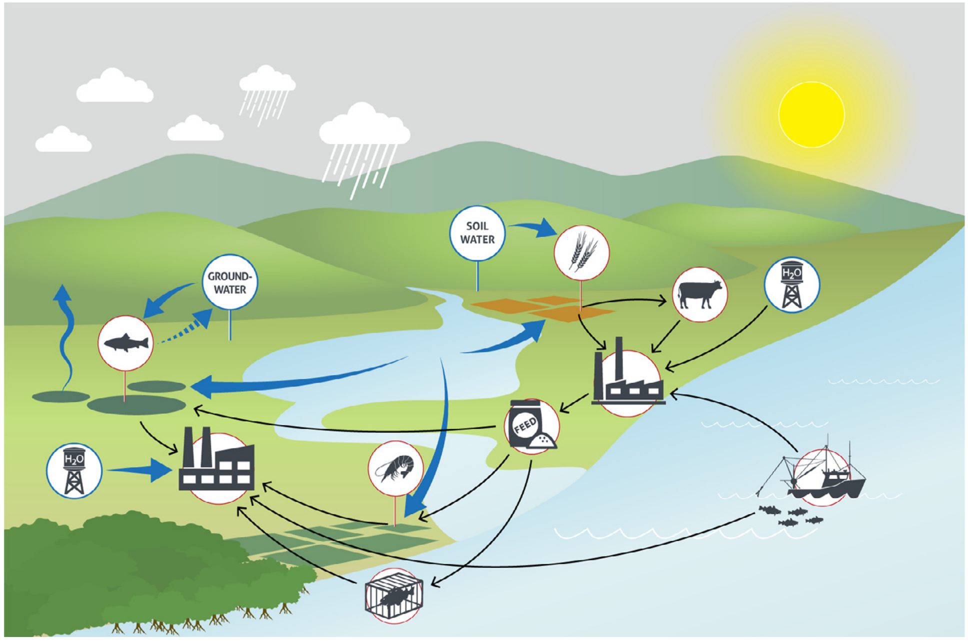

Fig. 1. Major water (blue lines) and product (black lines) flows in seafood production systems. Inland pond, coastal pond, cage, and capture fisheries are depicted, as well as fish meal/fish oil and aquafeed production. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

prints that, using the WFN approach, have been estimated to range from around 500 $m^3\ t^{-1}$ for gilthead seabream (*Sparus auratus*) and Atlantic cod (*Gadus morhua*), to more than 2500 $m^3\ t^{-1}$ for silver barb (*Barbonymus gonionotus*) (Pahlow et al., 2015). Going forward, the dependence on aquafeeds is increasing with intensified production systems and the composition of feeds is shifting (Tacon et al., 2011; Troell et al., 2014b; Fry et al., 2016). Due to sustainability concerns for capture fisheries which produce fishmeal and fish oil for aquafeeds (Cao et al., 2015), aquaculture is currently reducing its reliance on capture fisheries by incorporating more terrestrial, crop-based feed (Bell and Waagbo 2008; Beveridge et al., 2013; Fry et al., 2016). This shift will likely increase the indirect water footprint of aquaculture (Gephart et al., 2014; Troell et al., 2014b; Pahlow et al., 2015).

It is clear that some aspects of freshwater use in aquaculture, such as evaporation losses and reliance on feeds, map well onto water use in agriculture and livestock production systems. However, other aspects of water use for aquaculture production do not fit the same assumptions or do not have clear analogs in agriculture systems. As a result, we must reevaluate the assumptions and adjust the water use processes included in the analyses. Specifically, any analysis of water use for aquaculture should account for infiltration, timing and multiple uses related to storage and in situ or flow through water requirements.

Surface water diverted or groundwater pumped to fill aquaculture ponds experience variable infiltration rates depending on the geological characteristics of the underlying sediment and soil and the effectiveness of pond liners. Even if the water remains within the same watershed, infiltration processes change the flows and timing of water resources. For inland pond aquaculture, Verdegem and Bosma (2009) report average water infiltration of 7 $m^3\ kg^{-1}$ fish. Infiltration is less substantial in agriculture systems. In fact, the widely used WFN database found infiltration to be negligible for rice (the most similar production system to pond aquaculture) and is therefore not included in water footprints (Chapagain and Hoekstra 2011). Infiltration from reservoirs used for irrigation may also be considered in agriculture systems, but would need to account for the multiple uses of the reservoir.

It should be noted that all calculations for water use related to evaporation and infiltration are site-specific and highly sensitive to yield and culture duration. Yield impacts the evaporative and infiltration water use attributed to a unit of product because given two identical ponds, the same total evaporative and infiltration losses would be divided among more product in a pond with higher yields (resulting in a lower water use per unit of product). Culture duration also impacts the total evaporative and infiltration water losses because the water surface area is exposed to evaporation for a longer period of time. Together these factors introduce large variability in water use calculations for species or production systems, but also represent an opportunity to improve water use efficiencies (Verdegem and Bosma, 2009). Such opportunities contribute to freshwater use in aquaculture being sensitive to water availability and whether water pricing is applied (Phillips et al., 1991). Notably though, intensification is often accompanied by increased mechanical aeration (which increases evaporation rates) and water exchange (Boyd and McNevin, 2015). While a similar issue arguably exists for on-farm water use for livestock production (e.g. cleaning water), this is a less dominant water use so the effect of yield changes is less dramatic.

Two related considerations for freshwater use in aquaculture are water storage in aquaculture ponds and accounting for multiple uses. Freshwater stored in ponds is exposed to evaporative losses and seepage for longer periods of time, and changes the local water stocks and timing of those stocks. Water use related to storage in agriculture systems (e.g. irrigation water held in reservoirs) has generally been left for future research, but the analogous situation for hydropower has previously been studied (Mekonnen and Hoekstra, 2012b). Further, freshwater stored in aquaculture ponds that is not lost through evaporation or seepage is often released downstream, where it can be allocated for other purposes. Although a non-consumptive use, this makes that water unavailable for alternative uses during the storage period, and therefore represents a competitive use. However, if well timed, multiple water use can increase basin productivity, as has been shown in rice-fish production systems in Bangladesh (Nagabhatla et al., 2012). Unfortunately, temporal scales are hard to capture in all water footprinting methods, although some studies have considered temporal variability for crops (e.g. Pfister and Bayer, 2014; Kummu et al., 2014). Incorporating temporal scale can be important given the relevance to scarcity and the known inter- and intra-annual variability in water resources. For example, most aquaculture production is located in monsoon-affected areas, where water availability can go from droughts to floods within the same year. As a result, a high water footprint during times of high water availability may be misinterpreted as more problematic than a moderate water footprint during a drought. Aquaculture ponds may also act as freshwater reservoirs during dry seasons and large aquaculture sites may contribute to recharging aquifers.

Another important consideration for aquaculture is if and how one should account freshwater to open farming systems, such as cages, net-pens and reservoirs. Aubin et al. (2009), for example, accounted the water flowing through seabass cages towards the impact category "water dependence." This resulted in the water dependence of sea-bass being a thousand times larger than of trout in a flow-through open freshwater system. While this example is comparing freshwater and marine systems (which is not a meaningful comparison for freshwater use), the issue of water accounting for a cage system is still relevant. As mentioned above, water use by open systems is not consumptive use but is a competitive use and relates to water quality. In these systems, large volumes of non-consumptive water may be used to reduce the accumulation of persistent organic pollutants and to reduce the chances of disease outbreak when the replacement water is of better quality with respect to particles, oxygen, pests, and bacteria/viruses. However, in some cases reusing degraded water can increase the stress level of the animals and thereby trigger susceptibility to bacterial and virus diseases.

The above considerations for quantifying water use in aquaculture are highlighted through three case studies. First, we briefly present the water footprint of aquaculture in China using a method aligned with the WFN approach. In doing so, we illustrate the water use hotspots and briefly discuss regional scarcity. We then present results from an LCA-based study on water usage in Indonesian aquaculture and use this to illustrate the large difference among species and farming systems, as well as how methodological choices can influence the outcome. Later (Section 5.1), we present an aquaculture case from Egypt to illustrate issues related to water quality and the prioritization of water usage.

3.1. Chinese aquaculture: water use hot spots and regional scarcity

China produces more than one-third of the global fish supply and contributes more than 60% of global aquaculture production (FAO, 2016a). Freshwater aquaculture, especially of carps, tilapia, and catfish, dominates Chinese farming area and total production (FishStat, 2016). China reports its aquaculture production by body of water (i.e. ponds, lakes, reservoirs, river/ditches, rice-fish systems), by farming system (i.e. cages, pens), and water type (i.e. freshwater, brackish water, marine) (NBSC, 2012). This enables analysis of water use by specific aquaculture production system for China. We use the case of China to illustrate a water footprint calculation for seafood aligned with the WFN approach, use these results to illustrate water use hotspots, and briefly discuss this water use relative to scarcity.

The freshwater footprint of Chinese aquaculture was calculated following the methodology outlined in the Supplementary Information (SI), designed to align with the methods of the WFN (see Mateos, 2015 for a more detailed analysis). The footprint covers direct (on-farm) and indirect (off-farm) water use and includes blue and green water. The water footprint related to feeds was based on the water footprints of the agriculture components. Farm-made/local feeds are also used in Chinese aquaculture but this was

Table 2
Freshwater footprint of aquaculture ($m^3\ t^{-1}$) in China, disaggregated into the green and blue water footprints, and subordinated the most important water consuming processes. Ranges represent the range in estimates across provinces and species produced.

Type	Evaporation	Feed	Infiltration	Dilution	Total
Freshwater aquaculture					
Green	n/a	1361–2870	n/a	n/a	1361–2870
Blue	1067–16,621	67–603	724	n/a	1988–18,345
Total					3349–21,215
Brackish water aquaculture					
Green	n/a	1885	n/a	n/a	1885
Blue	n/a	313	n/a	218–5305* 6–54,927**	319–55,240
Total					2204–57,125
Marine					
Green	n/a	2495	n/a	n/a	2495
Blue	n/a	372	n/a	n/a	372
Total					2867

* River/Stream,
** Groundwater

not included in this analysis, which will result in an underestimate of the total water use. Low-value fish are also used as direct feed to Chinese aquaculture, but the freshwater footprint from this usage is negligible (Gephart et al., 2014) and therefore not included.

There are wide ranges in the blue and green water footprints for aquaculture in China that primarily stem from variation in the water footprints of feeds, production per area, eFCR, and dilution inputs (Table 2). The evaporative losses (ranging from 1067 to 16,621 $m^3\ t^{-1}$) contributed the most to the variation in the freshwater aquaculture water footprint, while the dilution inputs (6–54,927 $m^3\ t^{-1}$ for groundwater and 218–5305 $m^3\ t^{-1}$ for river water) contributed the most to the variation in the brackish pond water footprint. However, both the evaporation rates and dilution factors per tons of product varied across the regions due to a combination of environmental conditions and differences in yields. Thus, the evaporation itself did not differ so much between areas and farms, but the production varied by an order of magnitude, which introduced high variability in the evaporation and dilution expressed per unit of product. Additional details on water use for seafood production by region is available in the SI. These results indicate the water footprint of seafood in these systems could be reduced through designs minimizing evaporative losses and yield improvement measures.

The blue and green water use associated with the feeds varied due to the feed composition, species feed dependency, and eFCRs. The feed-associated water footprints were 1428–3473 $m^3\ t^{-1}$ for freshwater aquaculture, 2198 $m^3\ t^{-1}$ for brackish water aquaculture, and 2867 $m^3\ t^{-1}$ for marine aquaculture (Table 2). Details on the composition and origin of the feeds are provided in the SI. These estimates fall in the same range as the estimated global combined average green and blue water footprint associated with feeds in aquaculture (1808 $m^3\ t^{-1}$) (Pahlow et al., 2015).

The feed-associated water footprint of Chinese aquaculture is similar to the estimated global average (blue and green only) for chicken (2721 $m^3\ t^{-1}$), but well below that of pig meat (3545 $m^3\ t^{-1}$), and beef (9636 $m^3\ t^{-1}$) (Mekonnen and Hoekstra, 2012a; minimally-processed carcass values selected). This agrees with estimates of the feed-associated water footprint for seafood production, which found the water footprint of seafood to be less than that of terrestrial meat products on average, in terms of the mass and nutritional content of the products (Pahlow et al., 2015; Gephart et al., 2016). While only considering the water used for the feed production and the food conversion ratio is consistent with the boundaries of the WFN database approach to estimating the livestock water footprint, it misses the water consumption associated with evaporation, seepage, and dilution. This case illustrates that the evaporative- and dilution-related water use substantially increase the water footprint estimates and underscores the importance of expanding the processes included in the water footprint quantification methods when comparing terrestrial food to aquaculture.

In China, the most severe water scarcity is found in the North China Plain, in the Yellow and Yongding River basins (Hoekstra et al., 2012; Zhao et al., 2015). This region has been responsible for half of all wheat and one third of all maize produced in China (Kendy, 2003). These areas have historically not been used for aquaculture, but between 1979 and 2002 aquaculture grew at an annual growth of 15% (NASO, 2005). Still, most aquaculture production originates in the Yangtze and Pearl River basins (Wang et al., 2015) and these basins have among the lowest water scarcities. Currently the highest freshwater-aquaculture producing provinces (e.g. Guangdong, Juangsu, Hubei, etc.) are situated in the tropical and sub-tropical regions, where water scarcity is not yet an issue (Wang-Erlandsson et al., 2014). Nevertheless, monitoring both the water scarcity status and sectoral water use, including where feed resources originate, are important to ensure joint food-water security.

3.2. Indonesia aquaculture: water use by species and farming system

The aquaculture output of Indonesia has almost quadrupled over the last decade (FAO, 2016a). The majority of production comes from freshwater, primarily finfish grown in ponds, but also floating cage nets on rivers and lakes. Indonesia has an abundance of fresh water, with climate zones ranging from warm temperate moist to tropical wet. However, the largely unregulated expansion of aquaculture across the Indonesian archipelago has resulted in extensive water degradation in many coastal areas, with inhibited growth and fish mortality as a consequence (Mungkung et al., 2013). Logging and palm oil plantations have further degraded coastal water quality (Mukherjee and Sovacool, 2014), seasonally threatening aquaculture farms. Thus, the link between aquaculture and freshwater is important in Indonesia.

Phillips et al. (2015) compared freshwater consumption in different Indonesian aquaculture systems. Freshwater consumption included evaporation from freshwater ponds, freshwater diluted in brackish water ponds and freshwater consumed to provide supporting services (e.g. agricultural irrigation water or water used in refineries). Data were derived from the ecoinvent v2.2 database and Mekonnen and Hoekstra (2011). The results showed that dilution of marine water in brackish water ponds by far accounted for the largest consumption of freshwater (Fig. 2). Among the freshwater finfish farms, evaporation ranged from nothing in lake-based cage systems (tilapia and carps), to 30 and 71% of the freshwater consumed in *Pangasius* catfish pond systems using mass and economic allocation, respectively. The discrepancies between the two allocation factors are largely related to the water footprint associated with agricultural by-products, especially rice bran. However, dispersions of inventory data (evaporation rates, eFCRs, stocking density, fuel use, etc.) also resulted in almost an order of magnitude uncertainty. The marine cage farm systems had the lowest freshwater consumption, as they depended almost exclusively on marine farming environments and mainly used low-value fish as feed. For these systems, freshwater consumed in the oil production stage could account for up to 62% of the water footprint (but still have a low total freshwater consumption).

Unlike Phillips et al. (2015), and as argued against earlier in the text, Mungkung et al. (2013) did account water dependence to the grow-out part of tilapia and carp farming in reservoirs

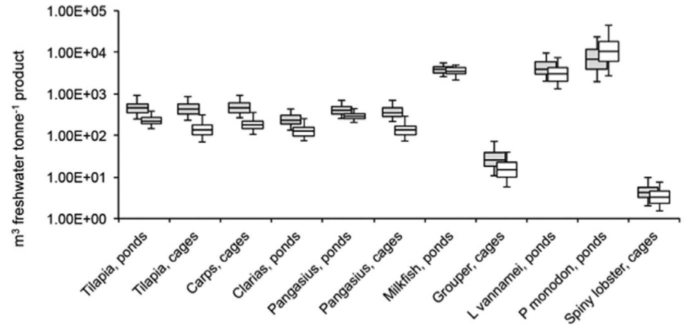

Fig. 2. Freshwater consumption in Indonesian aquaculture systems plotted against a lognormal scale. Grey boxes indicate results using mass allocation; white boxes results using economic allocation; the box indicate the median, the 25th and the 75th percentiles; whiskers indicate the 5th and 95th percentiles. Data from: Phillips et al. (2015).

by quantifying the water flowing into cages, as first proposed by Aubin et al. (2009). Nonetheless, they concluded that only 20% of the overall water dependence was related to the grow-out phase as a result of the low water renewing rate within the reservoir and high stocking density (Mungkung et al., 2013). Moreover, in absolute terms, the freshwater use estimates of Mungkung et al. (2013) using economic allocation were 7–8 times larger than those of Phillips et al. (2015), while the difference between mass and economic allocation resulted in 2–3 fold difference in outcomes. This demonstrates the importance of methodological and inventory choices which impact comparisons between products (Henriksson et al., 2015).

4. Water use in capture fisheries

Capture fisheries and aquaculture contribute equally to global seafood production. Capture fisheries require essentially no freshwater inputs apart from the relatively small water use associated with capture (e.g. water use in oil refineries), transportation (e.g. ice, and ship maintenance) and processing. While the consumptive freshwater use is negligible, a sufficient amount of water for living conditions must remain in fresh- and brackish water ecosystems to sustain the physical integrity, hydrogeomorphological processes and habitats the fisheries depend upon. For example, in addition to the freshwater volume requirements of organisms, reduced flows or lake levels may disrupt or destroy habitats for fish refuge and spawning (Brummett et al., 2010; Gaeta et al., 2014). This effect was observed during a prolonged drought in northern Wisconsin, when over 75% of previously submerged coarse woody habitat was lost from the littoral zone. During the same time, forage fish species fell below detection level and piscivore growth rates declined (Gaeta et al., 2014).

The volume of freshwater required to maintain inland fisheries is highly species and system specific and has been widely explored in the environmental flows (e.g. Loneragan and Bunn, 1999; Robins et al., 2005; Acreman and Ferguson, 2010) and water productivity (e.g. Brummett et al., 2010; Descheemaeker et al., 2013) literature. Through this work it is clear that although fresh and brackish water fisheries require freshwater to sustain them, the amount and timing of those water resource requirements must be assessed for each individual population. For example, a strong relationship was found between river discharge and fishery production in an Australian coastal fishery, but the impact differed substantially among species, with the response of prawn production to increasing flow being about twice that of mud crabs (Loneragan and Bunn, 1999). Additionally, in some cases the freshwater allocated to support fisheries is not available for alternate uses, such as agriculture, industry (e.g. cooling water, hydropower generation) or recreation, while in other cases the water can serve multiple uses (Fig. 1). This in situ water use and the fundamental issue of multiple uses of in situ water use is not well developed in the water use metric methodologies. While primarily a concern for capture fisheries and cage aquaculture, similar issues may arise when damming rivers to store irrigation water.

Complications in quantifying the volume of water required for fisheries is why capture fisheries are often associated with water area rather than volume (FAO, 2016b). Water areas better incorporate the underlying habitat complexity, can better account for systems where seasonal flooding is important, can incorporate multiple uses of the water, and are useful to delineate coastal habitats (FAO, 2016b). Despite the advantages of water areas, they do not provide a metric that is directly comparable to water used to produce other foods, both because of the unit differences (area versus volume) and differences in water use types (in situ versus consumptive). Whether using environmental flows or water areas, in situ water use estimates tend to be both variable and uncertain, which is problematic for water use quantification in systems where in situ water use dominates.

5. Water quality impacts

While the focus of this article is the quantity of water used throughout production, water quality impacts are also a concern. We recommend practitioners account for water quality using alternative impact categories or methodologies. In general, the WFN estimates degradative water use of agriculture based on the concentration of nitrogen released to the water and the quantity of water required to dilute to a baseline condition (grey water footprint; SI Table 1). The LCA community has also included water quality degradation in their grey water footprint. Bayart et al. (2014), for example, suggested quality to be included in their "Water Impact Index." In both cases, it is difficult to include the full range of water quality impacts, such as organic matter, biological activity, chemical interactions, or non-consumptive radiological impacts. Therefore, water quality changes should be accounted in separate impact categories, such as eutrophication and ecotoxicity, which can then be considered alongside water use. For more site-specific studies, risk assessment can account for temporal aspects, ecosystem thresholds, critical concentrations, 'cocktail' effects, and hydrological factors. As with water use quantity assessments, the scope of water quality assessments depends on the primary research question and the key point is to ensure the framework applied is inclusive of the most important processes in all production systems being considered.

Some primary water quality concerns for aquaculture differ from those in agriculture. For example, water quality impacts for aquaculture needs to account for a multitude of chemicals used on farm, such as disinfectants and antimicrobials. While antimicrobials are also applied in terrestrial livestock systems, the antimicrobials are delivered more directly to terrestrial animals. In both cases, the critical concentrations for disinfectants and antimicrobials released to the environment differ depending upon the organisms and systems affected (humans, freshwater vertebrates, marine invertebrates, etc.) and the degradation time may range from minutes to years. Our focus is on freshwater use, but there are similar water quality concerns in marine environments that could be evaluated for coastal and offshore aquaculture.

Another form of freshwater degradation not generally considered is groundwater salinization. This can be considered a quality issue, but also impacts the available freshwater by converting freshwater to saline water. Groundwater salinization is particularly relevant in coastal aquaculture systems. For example, coastal groundwater extraction to fill and maintain brackish water ponds has been linked to saltwater intrusion (Azad et al., 2009). Similarly, irrigation of crops using coastal water can also lead to groundwater salinization, but is not generally considered in water use analyses. As a result, when considering coastal aquaculture or agriculture systems, salinization impacts should be evaluated.

Beyond the stages of raising or catching seafood species, there is water use associated with processing, but this water use is primarily degradative rather than consumptive. According to Hall et al. (2011), different types of processing can result in vastly different water use. The authors, for example, highlight that surimi production uses more water than canning, which in turn uses more water than curing or freezing. Canning was estimated to require 15 m^3 t^{-1} raw materials processed, but the same quantity of water was estimated to leave the processing facility. Anh et al. (2010), similarly, conclude that all of the water used by pangasius fish processing plants producing frozen fillets (12.7 m^3 t^{-1} whole fish) is returned as wastewater, and 97% of the water (15 m^3 t^{-1} raw whole shrimp) from frozen shrimp processing plants. Some fish processing processes do consume water, especially when steam is required. For example, fishmeal processing has been estimated to result in losses of 0.6–0.7 m^3 t^{-1} raw materials processed as water vapor from boilers (Hall et al., 2011). However, in general the evidence suggests processing is more relevant to water quality degradation, rather than consumption.

5.1. Egyptian aquaculture: water quality considerations and water use priorities

Egypt is a warm temperate dry country with minimal precipitation. A study by Henriksson et al. (2017) concluded the freshwater footprint of Egyptian tilapia was between 10 and 42 times higher per ton compared to similar tilapia pond systems in Indonesia (see Section 3.2). The reason is that Egypt experiences a more arid climate and longer grow-out times (9 months) compared to Indonesia (3–4 months). As a result, between 88 and 92% of the freshwater consumed by Egyptian tilapia farming is associated with evaporation, compared to only 14–50% in Indonesia.

Most of Egypt's freshwater resources enter the country via the Nile River, after passing through ten other African countries to Egypt's south. The quantity and quality of water entering Egypt via the Nile is, however, increasingly influenced by the growing populations along the river's banks. Imbalances in the ratio of irrigation and drainage canals have further added to the problem and resulted in widespread salinity problems (Gad and Ali, 2011). This has led to a nationwide ban on the use of irrigation water in aquaculture, apart from exceptional cases (Khalil and Hussein, 1997). Most aquaculture farmers in Egypt are therefore largely limited to using poor quality agricultural drainage water with a salinity ranging from 0 to 3 ppt (Eltholth et al., 2015). Farmers generally respond to this by implementing large daily water exchanges. Further, since aquaculture farms in Egypt are the end users of the water they are the first to suffer from water shortages.

6. Data and methodological needs

Limited data is a major challenge for quantifying freshwater use for seafood production. Going forward, this requires: (1) improved geographic mapping of aquaculture sites; (2) improved data on stocking densities and yields; (3) improved data on aquafeed use, composition, and ingredient sourcing; (4) better models and maps for water seepage, including information on pond liners; (5) general models for calculating evaporation rates, covering both site-specific and larger geographical areas, and incorporating temporal variation; (6) additional analyses of environmental flows required to maintain capture fisheries, and; (7) water use data associated with processing. While there are crop maps with global coverage (e.g. IFPRI's Spatial Production Allocation Model and FAO's

Global Agro-Ecological Zones), little or scattered information is currently available on the location of aquaculture sites at the global level (even though such data exist for some countries, e.g. particularly for marine farming in Norway and Canada). There have recently been efforts to improve the mapping of aquaculture sites, such as FAO's National Aquaculture Sector Overview map collection (FAO, 2015) and Sea Around US database on Mariculture (Campbell and Pauly, 2012). Further, there has been a push for improved remote sensing tools to map aquaculture (see FAO, 2016a, p. 111, for discussion).

Going forward, quantifying water use for aquaculture will need to be developed first from local/ national perspectives to reach a regional and global assessment. A logical framework would be to first quantify water use by species and production system for the major producing countries. However, such an approach may overlook aquaculture practices in locations where water scarcity is already critical. Further, this is a tedious way of quantifying water use for the global sector, especially since getting real (and precise) quantification of the national production is already difficult (e.g. Metian et al., 2014) and it is more difficult to get production details within the country (few such initiatives exist: FAO or Sea Around Us project). This later aspect is key when coupling production with water use for which geographical (local to global) and temporal variation are essential.

Nevertheless, in order to overcome the lack of data at a global scale, rigorous approximations are usually the best approach (e.g. Mekonnen and Hoekstra, 2011, 2012a). Decisions need to be taken in order to facilitate the estimation of the water dependency of a sector rather than a real definition of it (*viz.* accurate quantification of it). In the context of the estimation of water uses in aquaculture, the biggest challenge in terms of data acquisition remains determining the quantity and the type of feed ingredients used in feeds and the quantity and the type of feeds used. For now, most global estimations are based on data collected in a survey done in 2010 (Tacon et al., 2011). It provides general information on the various ingredient inclusion rates but the geographical scale is limited and the time scale is absent. To address this, there is a need for more transparency from practitioners, particularly in terms of uses of feed ingredients and on-farm feed practices (e.g. Hasan and New, 2013). Such information remains difficult to obtain, especially when scientists/analysts are trying to assess a lucrative activity for which profits and productivity largely result from feeding practices. Future research should consider incentive structure options which can motivate information sharing between science and business to promote food-water nexus research.

Moreover, practices are widely variable and there are large differences among regions and across time, especially in small-scale production (Hasan et al., 2007), where evaporation from ponds can play an important role. In line with this consideration, responsible practices and improvement of technology modify ingredient use over time (Tacon and Metian, 2008). Thus, surveys have to be done on regular basis to see the dynamic aspect of the practices and thus the water uses. For example, FAO is collaborating with the salmon farming sector (Global Salmon Initiative) to learn lessons from how the sector successfully reduced average Food Conversion Ratios by 70% in three decades, thereby producing three times more fish per unit of feed, in order that similar approaches can be applied to farmed production of other farmed species.

7. Conclusion

Studying the food-water nexus requires a systems approach to the food sector, which includes seafood production. However, the seafood industry is highly diverse and not well represented as a single category in terms of its water use. For a start, water use differs greatly between capture and aquaculture systems. While aquaculture is similar to agriculture in some respects, some of the primary water use considerations differ due to the water storage requirements, which alter the evaporation and seepage. This introduces additional variability in water use estimates, as illustrated in the China case study. Next, the diversity of aquaculture production systems and differences in water accounting methods introduces another layer of variability, as illustrated in the Indonesia case. Specifically, past water use estimates for aquaculture span a huge range, from $1.5\,m^3\,t^{-1}$ for spiny lobster in Indonesia (Phillips et al., 2015), to $6\,m^3\,t^{-1}$ for generic farmed fish (Ranganathan et al., 2016), to $48{,}782\,m^3\,t^{-1}$ for sea-bass in netcages (Aubin et al., 2009). This also highlights a danger of comparing water footprints across studies, as modeling choices can outweigh data. Beyond methodological choices, one must also be careful about considering any type of water use versus water consumption, as illustrated in the Egypt case. With regards to production practices, marine systems reliant on wild fish or fishmeal generally perform the best. However, utilization of wild fish stocks in marine aquaculture feeds may aggravate problems of over-fishing (Cao et al., 2015). Such trade-offs, as well as connections of water use accounting to water scarcity measures, are important considerations.

Nevertheless, at a time of greater consumer interest in sustainable products, increasing data coverage and availability, and improving technology for tracking production, better water use accounting for seafood seems feasible in the coming years. The methodological considerations detailed above provide a foundation for utilizing this data as it becomes available. Until then, the range of previous research and cases presented here demonstrate the wide variation in water footprints among species and production systems. These methodological recommendations and findings represent an important step toward addressing the 'seafood gap' in the food-water nexus literature.

Acknowledgments

This work was funded by the Foundation for Strategic Environmental Research and The Swedish Research Council Formas. Jessica Gephart was funded in part by the National Science Foundation Graduate Research Fellowship Program and the National Socio-Environmental Synthesis Center under funding received from the National Science Foundation DBI-1052875. Thanks also to Vinnova Vinnmer Marie Curie Incoming (2015-01556) and Wenbo Zhang for support with data for the Chinese case study. The IAEA is grateful for the support provided to its Environment Laboratories by the Government of the Principality of Monaco. We also thank the two anonymous reviewers whose careful feedback helped improve and clarify this manuscript.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.advwatres.2017.03.025.

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