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Shifting from volume to economic value in virtual water allocation problems: a proposed new framework and methodology

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Abstract

Purpose: The water footprint provided a full methodology to operationalise the virtual water concept (the volume of water used along a supply chain to produce products and services). A key theme in the water footprint literature is the efficient allocation of water resources at the global scale given the feasibility of trading water intensive commodities from water rich to water poor areas: this is an economic problem of resource allocation between alternative and competing demands, albeit with a novel international component. Moreover, given that price signals indicating relative scarcity are usually either absent or distorted for water, it is also a problem that can be seen through the lens of environmental (or non-market) valuation. However, to date environmental valuation has not been used to inform the efficient use and allocation of water within and between the different locations encompassed by international supply chains.

Methods: Drawing on an agri-food supply chain framework which we propose in this paper, we begin by conceptualising the economic values that accrue to water consumption (blue and green water) and degradation (grey water) at different points along a supply chain. Based on this conceptualisation, we assess the extent to which it is possible to approximate these economic values by relying on existing secondary data on the shadow value of water in different contexts. The use of secondary data in this way is known as benefit (or value) transfer. To achieve this, 706 unit estimates of the economic value of water are collected, standardised and reviewed encompassing off-stream water applications (agriculture, industry and municipal) and in-stream ecosystem services (waste assimilation, wildlife habitat, recreation, hydrological functions and passive uses). From this, a proposed methodology for valuing virtual water is presented and illustrated using the case study of global durum wheat pasta production.

Results: The case study shows the total value of the virtual water used to produce one tonne of durum wheat pasta (\$212). More importantly, the case study also highlights how variations in economic value between multiple locations where durum wheat is cultivated (Saskatchewan \$0.10 m^3, Arizona \$0.08 m^3 and Baja California \$0.24 m^3) indicate relative water scarcity and thus impact, as well as the potential for a more efficient allocation of virtual water.

Conclusions: The main conclusion from this research is that when geographical disparities in the economic value of water use within a supply chain are accounted for, what was perhaps considered sustainable in volume terms, might not, in fact, represent the optimal allocation. However, future research opportunities

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highlight the need for additional data collection on the economic value of water in several contexts. This additional data would help the environmental valuation community to undertake a more comprehensive and robust approach to virtual water valuation.

This paper is accompanied by the Data in Brief article entitled "Dataset on the in-stream and off-stream economic value of water."

Keywords: Benefit transfer, stress-weighted water footprint, Total Economic Value, value of water, Water Footprint; water scarcity.

Abbreviations*:* AF, Acre Foot; AV, Average Value; BOD, Biochemical Oxygen Demand; CS, Consumer Surplus; ESS, Ecosystem Services; GDP, Gross Domestic Product; IPD, Implicit price Deflator; LCA, Life Cycle Analysis; MV, Marginal Value; PPP, Purchasing Power Parity; ROW, Rest of the World; TEV, Total Economic Value; USD, United States Dollar; WFA, Water Footprint Assessment; WTP, Willingness to Pay.

1. Introduction

Virtual water is the volume of water that is used along a supply chain to produce products and services (Allan 1996, 1998, 1999). As a concept, virtual water has shown how production in one location can impact water resources in distant geographies and the substantial water burdens that are often hidden within supply chains (Chapagain and Orr, 2008; Ercin *et al*., 2011).

The water footprint built on the concept of virtual water by developing a full methodology (Water Footprint Assessment or WFA) to account for different types of water use along supply chains (green, blue and grey) and their spatiotemporal distribution (Hoekstra, 2003; Hoekstra *et al*., 2011). The water footprint has been applied to individual products (e.g. Chapagain *et al*., 2006; Chapagain and Hoekstra, 2007; Chapagain and Orr, 2009; Chapagain and Hoekstra, 2011; Niccolucci *et al.*, 2011; Hadjikakou *et al.*, 2013; da Silva *et al.*, 2016) and the virtual water linked to consumption in specific geographies (e.g. Hoekstra and Chapagain, 2007; Van Oel *et al*., 2009), and it has been applied in industry (e.g. Jefferies *et al.*, 2012; Francke and Castro, 2013; Ruini *et al.*, 2013; Antonelli and Ruini, 2015) and policy contexts (Aldaya and Llamas, 2008; Aldaya *et al*., 2010). For a comprehensive overview, see Zhang *et al*. (2017).

At present, a principal and ongoing debate in the water footprint literature is taking place with the Life Cycle Analysis (LCA) community (Hoekstra *et al.*, 2009; Pfister and Hellweg, 2009; Hoekstra, 2016; Pfister *et al.* 2017; Zhang *et al*. 2018). The root of this debate centres on the purpose of the water footprint and – linked to this – how to conceive of a unit of water appropriation. Is the water footprint a tool to aid with the environmental impact assessment of products and their associated supply chains as suggested by LCA scholars? If so, should water volumes be weighted differently if they are consumed in an area of local water scarcity (Ridoutt, *et al.* 2009; Bayart *et al*., 2010; Ridoutt and Pfister, 2010; Kounina *et al*., 2013; Ridoutt and Pfister, 2013; Pfister and Ridoutt, 2014; Boulay *et al.*, 2015; Ridoutt *et al*., 2016; Ma *et al.*, 2018)? Alternatively, given that water is a global resource by virtue of the global economy (water intensive commodities can be traded between water rich and water poor areas), is the water footprint a means of assisting the optimum allocation of water volumes at a global scale, as advanced in the WFA literature (Hoekstra *et al.*, 2009; Aldaya *et al.*, 2010; Hoekstra, 2016)? If so, is the solution to the overexploitation of water not just to focus on water scare areas, but also to increase the productivity of water in water abundant areas? Furthermore, does every unit of water therefore have equal environmental relevance because its consumption reduces the availability of water for other purposes?

We will not rehearse this debate any further here. Nonetheless, the focus of WFA on the allocation of virtual water between alternative and competing uses at the global scale can be viewed as a classic economic problem of resource allocation, albeit with a novel international component. Moreover, given the unique set of characteristics associated with water that inhibit water management mechanisms such as markets, this focus is also particularly relevant to the sub-field of environmental or non-market valuation (henceforth environmental valuation) (Savenije, 2002; Hanemann, 2006; Young and Loomis, 2014). Environmental valuation focuses on applying welfare economic principles to estimate monetary values (shadow or accounting prices) for the goods and services provided by water. These estimates provide signals of the relative scarcity of water which would otherwise be lacking or distorted given the absence or ineffectiveness of markets. However, as Lowe *et al*. (2018) have pointed out, the academic discipline of environmental valuation has not overlapped with supply chain thinking and the virtual water and water footprint concepts. Environmental valuation has maintained a focus on deriving shadow values to ensure the most efficient use of a particular unit of water within a single water basin i.e. reallocating from lower to higher valued uses (e.g. Creel and Loomis, 1992; Loomis and McTernan, 2014). As a result, the *relative value* of *different units* of water across different geographies, as is the case along international supply chains, and the implications that this may have for the efficient allocation of water between basins, have not been addressed.

Attempts have been made to introduce economic-like concepts into the virtual water field by looking at *economic water productivity* along supply chains (e.g. Chouchane *et al*. 2015; Hogeboom and Hoekstra, 2017; Owusu-Sekyere *et al*. 2017; Miglietta and Morrone, 2018; Miglietta *et al*. 2018; Darzi-Naftchali and Karandish, 2019), or by using Input/Output modelling approaches that are founded on the concept of *value added* (e.g. Acquaye et al. 2017). However, neither approach can (or intends to) isolate the contribution that water makes. As a result, these approaches will not accurately estimate any shadow values attributed to water and thus are less helpful with allocation problems. In the non-peer reviewed grey literature, by contrast, high-profile tools have been developed that apply environmental valuation concepts to virtual water to estimate its ‘true’ economic value (PUMA, 2010; Høst-Madsen *et al.*, 2014a; Høst-Madsen *et al.*, 2014b; Ecolab and Trucost, 2015; Kering, 2015; Park *et al*. 2015; Ridley and Boland, 2015). These tools have been implemented by companies such as PUMA, Novo Nordisk and Kering. However, these tools have been developed as a means of understanding and managing supply chain risks rather than informing the allocation of virtual water, and the validity of the underlying methodologies has not been examined in the academic literature to date.

The primary aim of this paper is to assess the extent to which it is feasible to estimate robustly the economic value of virtual water. We undertake this assessment to show how economic value, with its microeconomic foundations in the concept of Willingness to Pay (WTP), can function as an indicator of relative water scarcity or impact, and thus water risk. However, we also suggest that the valuation of virtual water could improve spatial allocative efficiency and thus incentivise the efficient global allocation of virtual water.

The assessment is undertaken in the context of an agri-food supply chain framework that we use to set out what we mean by economic value and how this accrues to water appropriation in a supply chain. The agri-food focus of this framework has been chosen because agri-food supply chains provide the necessary degree of geographical variation without being overly complex, and because the agri-food sector both significantly impacts and is impacted by the availability of water (Ercin *et al*. 2011).

Given the geographical reach of modern supply chains and the demands that this would place on primary data collection, as well as the use of existing valuation tools such as ARtificial Intelligence for Ecosystem Services (ARIES) (Villa *et al*. 2014) and Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) (Sharp *et al*. 2015), this assessment is predicated on the use of existing data on the shadow value of water. This approach is known as benefit (or value) transfer in the welfare economics literature and involves transferring economic values estimated in one location (the study site) to a new location (the policy site). Benefit transfer methods range from single point value transfer where an estimate of WTP is transferred unaltered from the study site to the policy site (e.g. Prokofieva *et al.*, 2011; Kubiszewski *et al.*, 2013), to more advanced meta-value analysis which attempts to generate a pooled model from the results of several primary studies (e.g. Rosenberger and Loomis, 2000; Shrestha and Loomis, 2003; Bergstrom and Taylor, 2006; Brander *et al.*, 2012; Andreopoulos *et al.*, 2017). Benefit transfer is also the approach taken in the non-peer reviewed grey literature. However, unlike the approaches in the grey literature, the assessment presented here will take into account the widest possible range of water types (rainfall, surface and groundwater and water pollution) and water settings (off-stream and in-stream). In addition, it will be based on what the authors believe is the most comprehensive review of existing shadow value data conducted to date.[1]

Given that the subject of this paper is the development of a new methodology, it does not follow a conventional structure. In the first half of the paper we begin by presenting the proposed framework for valuing blue, grey and green water (explained below) along the supply chain (Section 2). Guided by this framework, Section 3 then outlines the methods that were employed to collect and standardise existing water shadow value estimates, and the results of this exercise are presented and discussed in Section 4. As this may indicate, the methods and results sections focus solely on the data collection exercise and not the development of the methodology that stems from this, which is the focus of the second half of the paper. In Section 5, we return to the valuation framework and assess its viability in light of the secondary data collected. As part of this, we present and illustrate a proposed new method for valuing virtual water in the context of a durum wheat pasta supply chain case study (Section 6). Finally, the conclusions highlight several suggestions for how this method could be developed and thus outline a tentative future research agenda (Section 7). In what follows, we refer to 'value' and 'economic value' interchangeably.

2. A Framework for the Valuation of Virtual Water

The framework presented here aims to capture the broad currents of economic value associated with water use in different locations along the supply chain. Local idiosyncrasies and variations in the timing of water availability and water quality are beyond the scope of what is proposed. As a result, the framework is best viewed as a tool for information gathering at low geographical resolutions, perhaps as part of an initial assessment of water use within supply chains. This qualification is particularly relevant given that secondary data will be used to estimate the economic values indicated by the framework as set out in Section 5.

Water use is understood here in traditional WFA terms with the recognition of blue, grey and green water (Hoekstra *et al*., 2011). Blue water refers to surface and groundwater. Grey water refers to the volumes of blue water that are necessary to assimilate pollutants. Green water is rainfall that is stored in the soil as moisture and utilised in agriculture and forestry. Green and blue water are accounted for in terms of water

[1] Indeed, as part of this review, all the assumptions that have been made will be disclosed, as will the nature of the economic values that are utilised, the exclusion criteria applied, and the means by which the values have been standardised and updated.

consumption, i.e. the volume of water that is no longer available at a particular space and time. For example, the water might have been incorporated into a crop during crop cultivation or evaporated in the course of industrial production.

Section 2.1 begins by describing the components of a typical agri-food supply chain and the different functional contexts in which water is employed. Sections 2.2 – 2.4 then conceptualise the economic value of the blue, grey and green water employed in these contexts.

2.1. Agri-Food Supply Chains

An agri-food supply chain is comprised of three distinct levels, and thus three distinct functional contexts in which water is employed, as shown in Figure 1. In reverse order, these levels include:

- Level 3 – An agricultural level whereby natural (green water) and artificial (blue water) irrigation are consumed during crop cultivation, and grey water volumes may be produced as a by-product of this.
- Level 2 – An industrial level that undertakes manufacturing and/or processing and which may consume blue water in the course of cooling, processing raw materials and general overhead requirements in factories. Again, grey water may result from these activities.
- Level 1 – A municipal or consumer use level if consumption of the product requires blue water for preparation, cooking or use.

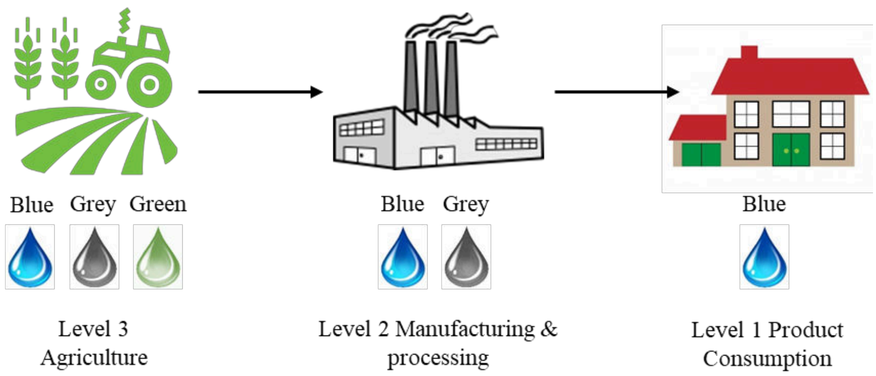

Figure 1. Conceptualisation of an agri-food supply chain showing functional variation between levels and associated green, blue and grey water. Blue water refers to surface and ground water. Grey water refers to the volume of blue water necessary to assimilate pollutants. Green water is rainfall that is stored in the soil as moisture and utilised in agriculture and forestry (Hoekstra *et al.* 2011). Black arrows indicate transition between levels of production and consumption.

Whilst the constituent parts remain the same, in reality, a supply chain may well include multiple agricultural, manufacturing and processing, and consumer use levels. Therefore, a challenge for the approach described here will be to go beyond the *functional* variations depicted in Figure 1, and robustly accommodate *geographical* variations in economic value as shown in Figure 2. Indeed, it is geographical variation, within each supply chain level, that has the potential to highlight the trade-offs between sites of functionally identical water use that an economic perspective draws attention to.

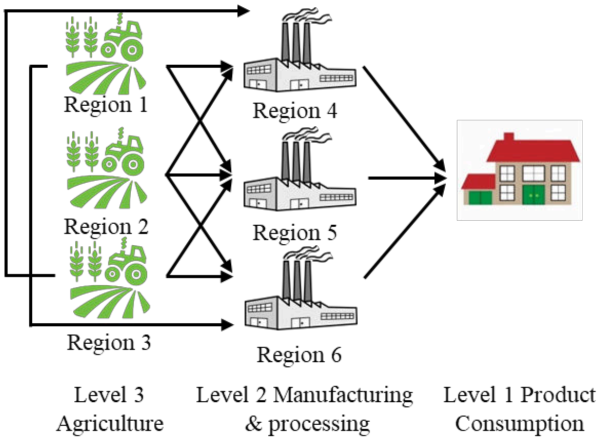

Figure 2. Conceptualisation of an agri-food supply chain including functional variation between levels and geographical variation between regions. Black arrows indicate transition between levels of production and consumption.

2.2. Conceptualising the Economic Value of Blue Water

The blue water component of the valuation framework is based on the idea that volumes of surface and groundwater have two values when employed at each supply chain level:

1. The value associated with the off-stream consumption of blue water in each of the functional contexts depicted in Figure 1 (i.e. agriculture, industry and municipal).[2] Off-stream water use involves extracting water from rivers, lakes and reservoirs and thus removing it from the natural hydrologic system.
2. The value associated with in-stream water services that are impacted when water is withdrawn and consumed in off-stream uses at each supply chain level. In-stream water services occur *in situ* and do not leave the natural hydrologic system.

To catalogue the range of in-stream water services impacted by consumptive blue water, the Common International Classification of Ecosystem Services (CICES) framework was utilised (Haines-Young and

[2] Given that some or all of these uses may be subject to a market price, it is being assumed here that existing prices paid for water in a supply chain may not have sufficiently internalised the full value of water.

Potschin, 2013). Table 1 sets out the five Ecosystem Services (ESS) selected from CICES that form part of the valuation framework. Each of the ESS shown in Table 1 has been subject to economic valuation to date, a consideration that also informed their selection (for example, see Gibbons, 1986; Frederick *et al.*, 1996; and Turner *et al.*, 2004). Other ESS based values could have been included in Table 1. For instance, in-stream values associated with hydropower and navigation, and functionally specific values applicable to wetlands. However, the decision has been made to exclude these values because of the low geographical resolution that has been chosen, as mentioned earlier.

Strictly speaking, the configuration of a water basin determines whether the value of in-stream water services can be added to the value of water extracted for off-stream use. In particular, this depends on the point of diversion (i.e. where water is diverted for off-stream use) and whether the in-stream water services are consumptive in nature (i.e. whether or not water is lost in their provision). If the in-stream services are non-consumptive (as they are in Table 1), then the value of a cubic metre of water consumed in an off-stream use is equal to the value of the full cubic metre in that use, plus the value of the in-stream ESS up until the point of diversion (Brown, 2004).[3] Given that these values are no longer in evidence when water is consumed, then the value of water can also be thought of as a cost (or dis-benefit).

As shown in Table 1, there is a correspondence between the ESS approach and the Total Economic Value (TEV) conceptual taxonomy proposed by Pearce and Turner (1990) that is commonly used to define the full range of values that are linked to the goods and services provided by natural resources. TEV includes direct use values, indirect use values and non-use or passive use values. Therefore, by estimating values for the range of in-stream ESS set out in Table 1, and off-stream uses that provide a direct use value, an approximation of TEV can also be derived. However, we are not suggesting that the nature of the demand for water in each of the three functional contexts in the supply chain encompasses the components of TEV. Clearly, off-stream water use in agriculture (Level 3) and industry (Level 2) is an intermediate input into production, and as such, it is subject to a derived demand (i.e. the demand is derived from the final good). Therefore, when TEV is mentioned in what follows, it is referring to the selection of off-stream uses and ESS

[3] The off-stream value would need to be an at source value net of input costs, such as pumping costs, to make it comparable with other in-stream values.

components, and not suggesting that the nature of the demand for water at any point along a supply chain encompasses all these components.

2.3. Conceptualising the Economic Value of Grey Water

Grey water is the volume of blue water required to abate pollution (Hoekstra *et al*. 2011).[4] Consequently, we will assume that grey water pollution impacts off-stream uses and in-stream ESS in the same way that the consumption of blue water does i.e. grey water takes the same value as blue water. However, unlike blue water consumption that physically deprives off-stream water uses and in-stream ESS of the associated volume of water, we recognise that grey water may still be available for some purposes, particularly in agriculture. For example, water polluted with Nitrogen and Phosphorous can still have a fertilisation effect when it runs-off cropland. Nevertheless, excessive Nitrogen and Phosphorous (or its mistimed application) can also impede crop growth. The contamination of run-off with, for example, pesticides and heavy metals can have a similar outcome. Given this uncertainty, the value assigned to grey water here is best conceived as an upper bound estimate.

In a similar way to blue water consumption, the value of grey water is based on the uses that this water could have been put to if it had not been impaired. Therefore, the value of grey water can also be thought of as a cost (or dis-benefit).

2.4. Conceptualising the Economic Value of Green Water

Green water does not impact in-stream ESS like blue, and as conceived here, grey water. However, green water nonetheless still has a value when it is consumed in, and thus supports, crop production at Level 3 in the supply chain.

[4] Given that economic valuation can only be applied to actual as opposed to theoretical volumes of water, to assign a value to grey water, it will be necessary to assume that there is not more pollution than assimilative capacity in the receiving water body. Liu *et al*., (2012) have examined historical and future trends in grey water associated with nitrogen and phosphorous discharges. They provide guidance on which global river basins this assumption is likely to be appropriate.

Table 1

Ecosystem Services selected from CICES that are impacted by blue water consumption

CICES Section	CICES Division(s)	CICES Group(s)	CICES Class(s)	Summary label	Summary Definition	TEV category
Regulation and maintenance	• Mediation of waste • Maintenance of physical, chemical biological conditions	• Mediation by ecosystems • Water conditions	• Filtration/sequestration/storage/accumulation by ecosystems • Dilution by atmosphere, freshwater and marine ecosystems • Chemical condition of freshwaters	Waste assimilation	The benefit provided by water bodies and rivers that dilute waste and thereby decrease damages that may be suffered by other water users.	Indirect use
Regulation and maintenance	• Mediation of flows	• Liquid flows	• Hydrological cycle and water flow maintenance • Flood protection	Hydrological functions	The capacity to alleviate floods and foster groundwater.	Indirect use
Regulation and maintenance	• Maintenance of physical, chemical biological conditions	• Lifecycle maintenance, habitat and gene pool protection	• Maintaining nursery populations and habitats	Wildlife habitat	The role that water plays in providing a habitat for fish and other wildlife.	Indirect use
Culture	• Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	• Physical and experiential interactions	• Experiential use of plants, animals and land-/seascapes in different environmental settings • Physical use of land-/seascapes in different environmental settings	Recreation	The benefits provided by direct access to water (e.g. rafting, kayaking and fishing), as well as shoreline based activities (e.g. camping and hiking) which are enriched by proximity to water.	Direct use
Culture	• Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes	• Other cultural outputs	• Existence and bequest.	Passive use	The benefit that stems from knowing that water resources exist and are available for current and future generations to use and/or exploit.	Passive use

Note. CICES Section, Division, Group and Class descriptors have been taken from Haines-Young and Potschin (2013).

It should be remembered here that green water is not rainwater; it only refers to that portion of rainwater that is evapotranspired by the crop (i.e. the portion that is consumed).[5] As such, the value of green water could be assumed equal to the value of artificial irrigation consumed by the crop. To make this assumption, however, it is also necessary to assume that the productivity of a unit of evapotranspired water is the same irrespective of timing (i.e. that there is a linear relationship between value and evapotranspiration levels). In reality, this relationship will vary with crop variety, and thus the suitability of this approach will be dependent on the context (Steduto *et al*., 2012).

Figure 3 provides a diagrammatic overview of the valuation framework adopted. As shown, in each of the three functional contexts along the supply chain, the value of blue and grey water is made up of: 1) the value associated with each off-stream application, and 2) the value of the in-stream water services impacted by the off-stream use. Green water is only utilised at the agricultural level and does not impact in-stream water services.

The viability of all aspects of the framework depends on the availability of corresponding data for benefit transfer (i.e. the existence of relevant shadow value estimates), a subject to which we now turn.

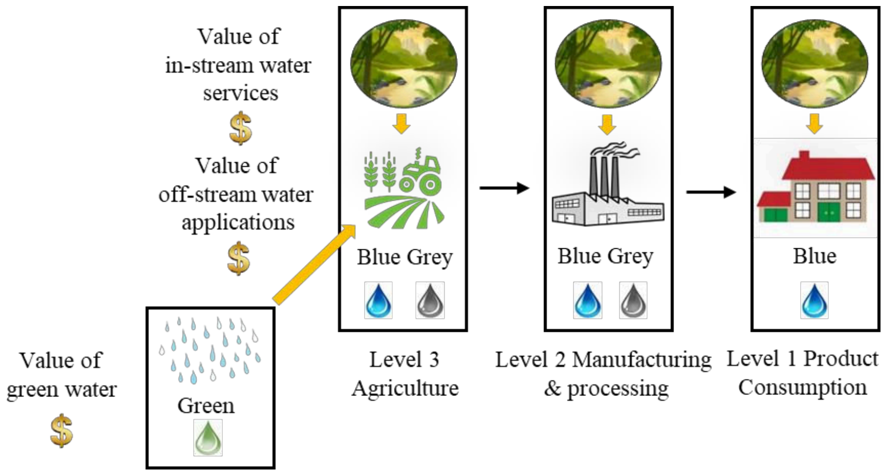

Figure 3. Proposed virtual water valuation framework including the value of off-stream, in-stream and green water. Blue water refers to surface and ground water. Grey water refers to the volume of blue water necessary to assimilate pollutants. Green water is rainfall that is stored in the soil as moisture and utilised in agriculture and forestry. Black arrows indicate transition between levels of production and consumption. Orange arrows indicate how the different water values feed into one another.

[5] If the aim had been to value rainwater more broadly, then the economic value associated with this would potentially have been negative depending on the time of year. For example, excess rain can lead to waterlogging which impedes crop growth.

3. Materials and Methods

Having introduced the valuation framework, this section provides an overview of the methods that were used collect and standardise the economic value estimates that correspond to this framework. The Data in Brief article that accompanies this paper provides a more detailed account of these methods and should be read in conjunction.

3.1. Cataloguing Existing Water Values

To compile the literature for this analysis, a detailed search of the five principal environmental valuation databases (Table 2) was conducted to identify relevant economic value estimates for the off-stream water uses (agriculture, industry, and municipal), and the in-stream ESS set out in Table 1.[6] The databases do not follow a common structure or reporting format and range from simple spreadsheets (The Economics of Ecosystems and Biodiversity Valuation Database and ValueBase SWE) to more sophisticated online interfaces (Environmental Valuation Reference Inventory) that allow a greater range of user enquiries. Economic value estimates denominated in volumetric units were sought so that they could be combined with specific volumes of water along a supply chain. [7]

The reference sections of identified sources were also checked for additional relevant material. In all cases, the sources identified in the search were consulted directly to examine the value estimates in detail. For a limited number of sources, the original data were no longer available, and thus a secondary reference had to be relied upon. Secondary references were only used if sufficient details were provided about the economic value and how it had been estimated originally.

Sources were excluded where: 1) they were not published in English, 2) they referred to one-off unit value estimates for water but with little associated explanation about how this estimate was derived, 3) they used non-standard volumetric units of measurement (e.g. a bucket of water), 4) they did not explicitly derive a unit value estimate but where this may have been feasible with sufficient knowledge of the original study and original context, and 5) they adopted a social as opposed to private accounting stance.[8]

[6] Three of the databases provided appropriate values for inclusion.
[7] Economic value estimates are denominated in various units, e.g. per acre, per household, per day.
[8] A social accounting stance involves adjusting any market prices used in the calculation of the water value for government subsidies and/or taxes (Young and Loomis, 2014, p.52).

Table 2

The environmental valuation databases consulted for this study

Database name	Supported/developed by	Publication date range of studies included	Approximate number of total publications included [a]	
			Pre-2000	Post-2000
Environmental Valuation Reference Inventory (EVRI)	Environment and Climate Change Canada and the UK Department for Environment and Rural Affairs.	1971 – 2019	1,386	3,552
ValueBase SWE	Beijer Institute for Ecological Economics and the Swedish Environmental Protection Agency.	1974 – 2003	110	60
Envalue	New South Wales government (Australia).	1969 – 2000	416	6
The Economics of Ecosystems and Biodiversity (TEEB) Valuation Database	Foundation for Sustainable Development, and Wageningen University.	1974 – 2010	465	845
The New Zealand Non-market Valuation Database	Lincoln University (New Zealand).	1973 – 2010	76	80

Note. [a] As at April 2019. Includes sources that focus on environmental goods and services other than water.

Although 120 publications ended up providing data for this exercise, nine publications in particular proved to be helpful in identifying relevant material (Young and Gray, 1973; Gibbons, 1986; Loomis, 1987; Colby, 1989; Brown, 1991; Frederick *et al.* 1996; Postel and Carpenter, 1997; Turner *et al.* 2004; Aylward *et al.* 2010).[9] In the case of Gibbons (1986), Frederick *et al.* (1996) and Aylward *et al.* (2010), these studies were also compilations of various value estimates, albeit they were more restricted in scope either owing to their age (Gibbons, 1986; Frederick *et al.* 1996) or explicit aims (Aylward *et al.* 2010). As can be seen, some of these papers are now outdated. This theme will be returned to in what follows.

3.2. Value Standardisation

In line with the approach adopted by Frederick *et al.* (1996) who among others (e.g. Rosenberger and Loomis, 2001) attempted a similar exercise to this, all economic value estimates are temporally adjusted to 2014 US Dollars (USD) using the Implicit Price Deflator (IPD) for GDP from the USA's Bureau of Economic Analysis (BEA, 2016). Where economic values were denominated in currencies other than USD, following the approach advocated by Ready *et al.* (2004) and Czajkowski and Ščasný (2010), they were first converted into USD using World Bank Purchasing Power Parity (PPP) exchange rates for GDP applicable to appropriate valuation year, before being temporally adjusted to 2014 using the IPD (World Bank, 2016).

[9] Note the paper by Aylward *et al.* 2010 was discovered separately to the main literature search.

Where values were given as a simple range (e.g. $10 - $20), the median value was used in the standardisation procedure. Where a value was listed as greater than a certain figure (e.g. >$100), then the value given (in this case $100) was used. Where the source provided multiple estimates in the form of a marginal relationship (e.g. marginal recreation values according to different levels of water flow) then the median value in the range (and the range itself) has been recorded to ensure that the value is one that is observed. This has been necessary because there are multiple estimates, across different value categories, which have been derived using a variety of different variables, not all of which can be considered. However, as presented in Table 3, several subcategories have been defined within each water category to classify the data.

Table 3
Sub-categories used to classify the economic value estimates

Water category	Sub-categories
Agriculture/Irrigation	Per period or capitalised asset. At site or at source. Short-run or long-run. Water measure (withdrawal, application or consumption). Crop value (low and high).
Industry	Sector
Municipal	Domestic specific (Y/N). Per period or capitalised asset.
Waste assimilation	Pollutant type. Point or non-point pollution.
Wildlife habitat	Per period or capitalised asset. Wildlife type.
Recreation	Flow variation. Recreational activity. Site characteristics.

Given that the majority of the economic value estimates were USA specific (nearly 60%), and thus denominated in Acre-Feet (AF), this was the standardised volumetric measure used to summarise the data so as to minimise the number of conversions required (1 AF = 1,233.48 m^3).

While every effort has been made to standardise the economic value estimates, the original calculations were made by many different authors who have used a variety of market and non-market valuation methods. These methods include cost-based techniques that are not based on the demand curve. Stated and revealed preference techniques (that provide genuine welfare estimates either in terms of Marshallian consumer surplus or the Hicksian compensating or equivalent measures) are also included. As a result, some of the estimates are not directly comparable in a strict sense. Indeed, some of the techniques used to generate the value estimates give rise to average values, some give rise to marginal values, and others derive the average value of a marginal increment. In some cases, it is not possible to identify what value conception is being identified, as the authors do not always make this explicit. This methodological variation has been addressed in the accompanying Data in Brief paper by ensuring that the value estimates can be broken down by valuation technique as well as by geography. However, given this unavoidable variation, the summary statistics that we move on to should be considered broadly indicative only; much of the variation in the data will only be fully discernible by directly consulting individual data points in the associated Data in Brief paper.

4. Results

In this section, we present an overview of the results from the detailed literature search. The Data in Brief paper that accompanies this article provides every data point captured.

4.1. Overview

The search yielded 706 unit estimates of economic value, across 120 different sources that were authored between 1956 and 2015. The sources included journals, books, working/discussion papers, reports and theses. The economic value estimates have been divided into two groups, which reflects a deep geographical division found in the literature:

- Those that refer to the USA – 408 estimates (or 58% of total estimates) from 69 sources (median publication date 1985).
- Those that refer to the Rest of the World (ROW) – 298 estimates (or 42% of total estimates) from 53 sources (median publication date 2005). Two sources were common to the USA and ROW and are therefore included in both groups.

Table 4 presents the number of economic value estimates by category; the capitalised asset values collected (46 in total) are included in the accompanying Data in Brief paper but have been excluded here as they are not relevant in this context.[10] Unit value estimates were available for all of the off-stream uses, and three of the five in-stream ESS set out in Table 1 (waste assimilation, wildlife habitat and recreation). However, no suitable hydrological values for use in this context were discovered. In addition, there was only one study on passive use values that was denominated in unit value terms (Loomis, 2012). Therefore, it was not feasible to include hydrological or passive use values in the analysis that follows.

[10] Capitalised asset values represent the capitalised present value of a stream of future values attributable to water.

A summary of the economic value of water in each of the off-stream and in-stream uses set out in Section 2 is addressed in turn below. Section 5 will then move on to discuss how these value estimates can be deployed in the context of a supply chain.

Table 4
Number of estimates and sources by category

Category	Number of estimates	Number of sources
Agriculture USA	209	34
Agriculture ROW	144	35
Industry USA	42	10
Industry ROW	89	9
Municipal USA	25	6
Municipal ROW	65	18
Recreation USA	49	27
Waste assimilation USA	13	6
Wildlife habitat USA	24	7

Note. USA = United States of America. ROW = Rest of the World.

4.2. Off-stream Values

Tables 5 and 6 summarise the agricultural values from the USA (209 estimates from 34 sources) and ROW countries (144 estimates from 35 sources). These estimates reflect the value of artificial irrigation water. As shown, the value of irrigation water can be defined by the measure of utilisation i.e. the volume of water that is withdrawn or diverted from a water source, that which is applied to the crop, or, that portion of applied water that is consumed during crop growth (sometimes referred to as net irrigation). The value of irrigation water can be further defined in three ways: 1) at the source of water extraction or at the site where it is used (depending on whether any costs incurred in extracting the water from the stream and making use of it are included when deriving the water value), 2) in the long and short-run (depending on whether or not fixed costs are taken in to account when deriving the water value), and 3) for high value (or speciality) or low valued crops.

Most of the irrigation values from the USA came from the south and west of the country, a feature that is mirrored in the other value categories presented here. The values for the ROW encompass 21 countries (Australia, Canada, Cyprus, Egypt, Greece, India, Indonesia, Iran, Jordan, Kenya Mexico, Mongolia, Morocco, Pakistan, South Africa, Spain, Sri Lanka, Tanzania, UK, Ukraine and Zimbabwe).

Table 5

Agricultural water values (USA) by type

	Number of estimates	Mean (2014 $/AF)	Median (2014 $/AF)	Standard deviation
All estimates USA	209	105.41	65.02	167.69
Location				
At site	152	113.83	74.45	175.32
At source/in-stream	32	68.63	52.73	61.75
Short/long-run				
Short	66	110.75	95.51	90.28
Long	86	67.80	58.86	57.11
Volumetric measure				
Withdrawal	18	45.54	21.78	58.23
Application	147	121.81	80.93	176.23
Consumption	20	34.28	26.25	21.76
Crop value				
High	49	152.86	134.88	88.40
Low	94	112.63	65.90	208.45

Note: USA = United States of America. $ = United States Dollar (USD). AF = Acre Foot. Values converted to 2014 USD using the Implicit Price deflator for GDP from the Bureau of Economic Analysis (BEA, 2016). At site = the value of water at the site of use. At source = the value of water at its source (or in-stream). Short run: the value of water after accounting for variable costs. Long run: the value of water after accounting for fixed and variable costs. Water withdrawal = the volume of water that is withdrawn from a surface or groundwater source. Water application = the quantity of water that is delivered to the location where it will be used. Water consumption = the volume of water that is no longer available at a specific place and/or time because it has been lost, for example during the process of evapotranspiration (by crops, trees etc.). High value = crops classified as high value (or speciality crops) because they produce high annual income per unit of land (e.g. fruit). Low value = all other non-speciality crops. Classification of crops by value based on El-Ahry and Gibbons (1988).

Table 6

Agricultural water values (ROW) by type

	Number of estimates	Mean (2014 $/AF)	Median (2014 $/AF)	Standard deviation
All estimates ROW	144	595.94	148.44	1,752.22
Location				
At site	51	309.28	180.94	403.95
At source/in-stream	19	217.11	143.45	246.23
Short/long-run				
Short	46	184.91	90.96	236.02
Long	11	60.25	37.94	51.19
Volumetric measure				
Withdrawal	7	177.12	143.45	158.67
Application	67	531.96	150.63	1,241.81
Consumption	20	973.47	479.03	1,394.74
Crop value				
High	13	2,644.70	905.46	4,673.40
Low	65	486.45	173.70	1,091.98

Note: ROW = Rest of the World. $ = United States Dollar (USD). AF = Acre Foot. Values converted from local currency to 2014 USD using World bank PPP exchange rates for GDP and the Implicit Price deflator for GDP from the Bureau of Economic Analysis (World Bank, 2016; BEA, 2016). At site = the value of water at the site of use. At source = the value of water at its source (or in-stream). Short run: the value of water after accounting for variable costs. Long run: the value of water after accounting for fixed and variable costs. Water withdrawal = the volume of water that is withdrawn from a surface or groundwater source. Water application = the quantity of water that is delivered to the location where it will be used. Water consumption = the volume of water that is no longer available at a specific place and/or time because it has been lost, for example during the process of evapotranspiration (by crops, trees etc.). High value = crops classified as high value (or speciality crops) because they produce high annual income per unit of land (e.g. fruit). Low value = all other non-speciality crops. Classification of crops by value based on El-Ahry and Gibbons (1988).

Across all estimates, the median value of irrigation water in the USA was $65 compared to $148 in the ROW countries. However, as shown in more detail in the accompanying Data in Brief paper, these values are not based on a like for like comparison. Indeed, the number of water values, how they break down by the sub-categories shown in Tables 5 and 6, and the methods for estimating the values, differs between the USA and ROW data pools. These differences make direct comparison of summary statistics difficult as variations other than geography may be driving the differences observed. In addition, as with all of the off-stream and in-stream categories summarised here, ROW currencies were converted to USD using PPP rather than nominal exchange rates which will have the effect of reducing the disparity between values denominated in USD and other currencies.

Table 7 summarises a selection of the industrial values captured in the literature search from the USA (42 estimates from ten sources) and ROW countries (89 estimates from nine sources). Those estimates generated by the added value, cost of intake and residual value approaches that are now considered inappropriate for valuing industrial water (Young and Gray, 1973; Gibbons, 1986), have been excluded here. However, these estimates have been included in the accompanying Data in Brief paper as they show how, what is a limited number of approaches to valuing industrial water, have evolved over time. Nonetheless, industrial values generated by the added value, cost of intake and residual value approaches should be treated with caution, as the artificially high nature of these values in many cases suggests.

Table 7
Summary of industrial water values

Method	Number of estimates	Mean (2014 $/AF)	Median (2014 $/AF)	Standard deviation
Selected estimates USA [a]	18	173.75	21.31	373.07
Selected estimates ROW [b]	82	2,446.59	618.09	4,769.59

Note: USA = United States of America. ROW = Rest of the World. $ = United States Dollar (USD). AF = Acre Foot. Values converted from local currency to 2014 USD using World bank PPP exchange rates for GDP and the Implicit Price deflator for GDP from the Bureau of Economic Analysis (World Bank, 2016; BEA, 2016). [a] Alternative cost method only. [b] Alternative cost, cost function, input distance function and production function methods only.

As with agricultural values, industrial values from the USA are predominantly concentrated in the southern states. Industrial values from ROW countries include Canada, China, India, Mexico, Mongolia, and the Philippines. Across those estimates selected, the median value in the USA was $21, compared to $618 in the ROW countries. Again, however, this is not a like for like comparison as the techniques used to estimate these values differ between the USA and the ROW countries. In addition, the types of industrial use that the water is put to, and the water quality requirements associated with this, (which are the primary drivers of water value in this context), also differ between the USA and ROW countries.

Table 8 summarises the municipal values from the USA (25 estimates from six sources) and ROW countries (65 estimates from 18 sources). These estimates include the value of water used around the home, both indoors (e.g. cooking and hygiene) and outdoors (e.g. lawn sprinklers), as well as the value of water used in commercial (non-industrial) business activities.

Table 8
Summary of municipal water values

Method	Number of estimates	Mean (2014 $/AF)	Median (2014 $/AF)	Standard deviation
All estimates USA	25	230.83	91.96	257.22
All estimates ROW	65	1,752.07	482.83	4,251.70

Note: USA = United States of America. ROW = Rest of the World. $ = United States Dollar (USD). AF = Acre Foot. Values converted from local currency to 2014 USD using World bank PPP exchange rates for GDP and the Implicit Price deflator for GDP from the Bureau of Economic Analysis (World Bank, 2016; BEA, 2016).

The value estimates for the ROW countries encompass 14 mainly developing countries or territories (Canada, China, El Salvador, Honduras, India, Madagascar, Mongolia, Nicaragua, Nigeria, Palestinian Territory, Panama, South Africa, Tanzania and Thailand). Across all estimates, the median value of municipal water in the USA was $92 compared to $482 in the ROW countries. In part, this appears to reflect the fact that the value estimates from the USA that have been provided by water market transactions exhibit substantially lower values than those estimated using other techniques, thus impacting the overall summary statistics for the USA. However, the same caveats apply as noted previously with agricultural and industrial values.

4.3. In-stream Values

Table 9 presents a summary of the economic value estimates associated with the three in-stream ESS for which values were available. Given that all of the values discovered come from the USA, they are presented here together.

Table 9
Summary of in-stream values (USA)

	Number of estimates	Mean (2014 $/AF)	Median (2014 $/AF)	Standard deviation
Recreation	49	43.57	13.32	84.90
Waste assimilation	13	7.53	2.05	11.97
Wildlife habitat	24	59.67	55.61	42.65

Note: USA = United States of America. $ = United States Dollar (USD). AF = Acre Foot. Values converted to 2014 USD using the Implicit Price deflator for GDP from the Bureau of Economic Analysis (BEA, 2016).

The literature review discovered 49 estimates of the recreational value of water (median value $13) stemming from 27 separate sources. The majority of these estimates are for river-based recreation and have been derived using approaches that aim to establish the relationship between variation in the level of flow in a river (measured in cubic feet per second) and the associated marginal value.

Thirteen value estimates for waste assimilation were identified (median value $2), stemming from six different sources. For twelve of the thirteen estimates, value has been estimated using an alternative cost approach (waste treatment costs foregone); the remaining techniques estimated the value of the damages avoided. Pollutants analysed include Biochemical Oxygen Demand loadings (BOD), thermal pollution, and salinity. The majority of the waste assimilation values come from Meritt and Mar (1969) and Gray and Young (1974), which appear to be the only papers exclusively focused on the estimation of waste assimilation values. Both of these sources report value estimates at high levels of geographical abstraction (water basins or water regions), and as such provide limited detail on how the value of waste assimilation varies by geography. Moreover, both are now outdated, and they have not been improved. Indeed, the paper by Gray and Young (1974) (which appears to be a development of the earlier work by Young and Gray (1973)) was the only waste assimilation paper cited by Frederick *et al*. (1996) in their thorough review of the unit value estimates of water in the USA. It is also the only paper cited at any length by Gibbons (1986) in their review of a similar nature.

Twenty-four estimates of the value of water for wildlife habitat (median value $56), originating from seven sources, were identified. The majority of these estimates were derived from water market transactions and as such have been reported at high levels of geographical abstraction (e.g. at the US state level). Again, this provides limited detail about how wildlife values vary by geography.

5. Assessing the Viability of the Framework for Virtual Water Valuation

So far, we have presented a framework that could be used to value water along an agri-food supply chain (Section 2) and set out the methods used to search for, catalogue and standardise the valuation estimates in the literature that correspond to this framework (Section 3). In addition, we have summarised the range and geographical spread of these estimates (Section 4). We now assess the viability of the valuation framework considering the estimates collected in Section 4 and the benefit transfer techniques that these will permit, beginning with off-stream water uses.

5.1 Off-stream Values

Of the three off-steam uses examined (agriculture, industry and municipal), by far the greatest number of value estimates were discovered for artificial irrigation (agriculture). As such, the agriculture category holds the greatest potential for the use of advanced benefit transfer techniques (in particular predictive meta-value analysis models) to predict the economic value of artificial irrigation in a range of locations that a supply chain might span. For that reason, the 209 values recorded in the USA were selected to form the basis of a regression model.[11] This model was based on the theoretical framework set out in Scheierling *et al.* (2006) who conducted a regression on estimates of the price elasticity of irrigation water demand. However, the results from this exercise did not yield a useful predictive model owing to omitted variable bias – many of the sources analysed did not consistently comment on all of the variables Scheierling *et al.* (2006) suggested – and of those tested, it was only dummy variables that proved to be significant. Water stress (or scarcity) was also examined as a potential explanatory variable on its own given its use in approaches in the non-peer reviewed grey literature (e.g. Park *et al*. 2015). Although the use of water stress as a single explanatory variable is not grounded in an encompassing theoretical framework such as the production function used by Scheierling *et al.* (2006), it is not devoid of theoretical foundation given the link to basic demand and supply theory. Based on the available data though, the results indicated that water stress is not a predictor of the value of artificial irrigation.

Given the results from the regression modelling, it is clear that there is too much variation in the agricultural values category to make anything other than single point benefit transfer viable i.e. the transfer of individual estimates of WTP for artificial irrigation from the study site to the site of the supply chain level (policy site). These findings imply that only those geographies where a unit value estimate already exists, or neighbouring geographies with similar characteristics, can be covered by agricultural values collected here. However, even in these cases, three considerations are particularly important in this context: 1) primary study measurement errors, 2) generalisation errors, and 3) definition of a consistent scenario (Johnston and Rosenberger, 2010).

[11] Coming from a large and diverse country, utilising these values ensured that the data collected for the independent variables were available in a consistent format across the various subnational units.

On the first of these, Section 3 and the accompanying Data in Brief paper set out the criteria that were used to select and present the source material gathered here. These criteria ensure that the values collected had used appropriate and scientifically sound methods and were broken down by relevant sub-categories. In terms of the second point, generalisation error, the pool of agricultural values is limited in number and unevenly spread. As a result, tests of convergent validity on any transferred values are not feasible, and sensitivity analysis will instead be necessary to understand how sensitive conclusions are to changes in the agricultural unit values utilised. Indeed, given that transfer error is potentially magnified by focusing on the relative value of water between different locations, sensitivity analysis is particularly important. In addition, it should be noted that the level of precision sought in benefit transfer is a function of the significance of the policy decision (Johnston and Rosenberger, 2010). Therefore, given that the method here is looking to provide high-level information gathering, perhaps for the initial screening of supply chains, then a higher level of transfer error becomes acceptable. Where a transfer does not strictly occur (i.e. where a value exists for the geography of interest), consideration should still be given to the geographical scale of the value, with aggregate approaches providing economic values that are more representative of broader geographies, but field-level and crop-specific studies being far more numerous. Finally, regarding the third point, the definition of a consistent scenario is necessary to ensure that when agricultural values are being compared across geographies, as will occur when agricultural crops are sourced from multiple locations, the same object of valuation is being considered. In practice, this means ensuring that the irrigation water value type (at source/at site, long-run/short-run) is as similar as possible in each location to ensure that disparities such as these, as much as practicable, do not account for the divergences observed.

As well as seeking to value artificial irrigation in agriculture (blue water), the valuation framework presented in Section 2 also conceptualised a means of assigning an economic value to green water (Section 2.4). However, this relied on sufficient values being available that reflect the value of water consumed in agriculture. As only 20 such value estimates were discovered, it seems infeasible to assign an economic value to green water and thus this remains an outstanding research question of note.

Unlike agricultural values, the number of suitable estimates of the value of industrial water use was more limited. Indeed, if those estimates derived from alternative cost approaches that date from the 1970s were also excluded here owing to their relative simplicity, then there are only four contemporary studies that have focused on industrial water in a concerted way (Renzetti and Dupont, 2002; Wang and Lall, 2002; Kumar, 2004; Bruneau, 2007). Between them, these four sources value water in a wide variety of industries and sectors (e.g. chemicals, food and beverage, minerals, paper and paper products, petrochemicals, pharmaceuticals, power generation, and textiles) and cover industrial water use in developed (Canada and China) and developing countries (India). Therefore, these sources do provide some scope to estimate the value of water to industry in supply chains such as that depicted in Figure 1 i.e. where there is only one industrial location. However, geographically differentiated values, which would be relevant to a supply chain with multiple industrial sites of the same type (i.e. Level 2 in Figure 2), are not likely to be feasible. Neither, therefore, are the resulting trade-offs that such values would enable.

Given that the focus in the next section will be an agri-food supply chain case study, for illustration, Table 10 presents the economic value of water used in the food sector in Canada and China based on two of the four contemporary studies referred to above. These values are presented here per cubic metre as this is the volumetric measure of virtual water that will be used in the case study. Other values for other sectors of interest are also available in the accompanying Data in Brief paper.

Table 10

Food industry values

Source	Method	Value type	Water volume measure	Original value/m^3 (currency)	2014 $/$m^3$
Wang & Lall (2002)	Production function	MV	Consumption	2.57 (Yuan)	1.87
Bruneau (2007)	Alternative cost	AV	Consumption	2.5 (CAD)	2.92

Note. MV = marginal value; AV = average value; Yuan = Chinese Yuan. CAD = Canadian Dollar. m^3 = cubic metre.

The municipal value estimates introduced in Section 4.2 were limited in number (particularly in the USA). In addition, a large proportion of the estimates were derived from a range of rudimentary techniques. Therefore, this is a poor basis for any benefit transfer exercise. However, a standard equation for the integral of a demand function that was used by Young and Gray (1973) and Gibbons (1986) – who between them accounted for a large share of the USA municipal values – can be employed to estimate the value of water for domestic (i.e. residential) purposes in potential supply chains i.e. excluding non-residential municipal uses. The most accessible version of the equation comes from Young and Loomis (2014, p.238) and is set out below. However, Young and Gray (1973) and Young and Loomis (2014) both ascribe this expression to James and Lee (1971).

As shown, to approximate WTP (V), an estimate of price elasticity (E), a unit price (P), and an initial quantity (Q) are required. [12] The use of this equation to derive the value of domestic water use in the supply chain will be illustrated in the case study that follows.

$$V = [(P \times Q_1^{\frac{1}{E}}) / (1 - \frac{1}{E})] * [(Q_1^{1-\frac{1}{E}}) - (Q_2^{1-\frac{1}{e}})] \quad (1)$$

5.2. In-stream Values

Of the three in-stream water categories examined (recreation, waste assimilation and wildlife habitat), the greatest number of value estimates were available for recreation. Indeed, there were sufficient estimates available of the recreational value of water that meta-value benefit transfer could be attempted with those estimates that have established a relationship between water value and water flow. Nevertheless, the size and profile characteristics of the rivers covered in the respective studies would need to be controlled for given that low flow levels on one river might represent high flow levels on another, and vice versa. However, there are several problems with conducting such a regression analysis if the aim is to use this technique to predict recreation values in a range of disparate policy sites. For example, the intensity of flow at each supply chain level will be unknown at the level of spatiotemporal detail that is the focus here. Moreover, even if the intensity of flow were known, it is unclear if this should this be measured at the site of the supply chain level or the broader basin in which it sits. Linked to the previous point, the distance decay effect (how the value will decay or decline with distance from the recreational site) will also be unknown (Pate and Loomis, 1997; Hanley *et al*. 2003). In concert, these factors make it prohibitively difficult to estimate robust recreational values across geographies.

Given the limited coverage of the data points available for waste assimilation and wildlife habitat, and the fact that the majority of these data points were reported at high levels of geographical abstraction, it is also infeasible to estimate these value categories across geographies reliably.

6. A Proposed New Methodology

As we have seen, either because of limitations associated with the number of data points, or limitations associated with the available methods, it does not appear feasible to use benefit transfer to estimate

[12] Equation 1 represents the at site value of residential water. The at source value, equivalent to net consumer surplus (CS), can also be derived by subtracting average revenue from total WTP: CS = V – [(P_1) (Q_1-Q_2)] (see Young and Loomis, 2014, p.239).

in-stream values for a wide range of geographies that a supply chain might span. This conclusion casts doubt on those approaches in the non-peer reviewed grey literature that claim to be able to do so (PUMA, 2010; Ecolab and Trucost, 2015). In addition, it also means that the method presented here will not be able to approximate all of the components of TEV (Section 2.2.). However, we have also seen that it is possible, using Equation 1, to estimate the economic value of water in the domestic settings that a supply chain might encompass (Level 1). Similarly, it appears feasible to estimate functionally specific shadow values for water used in different industrial sectors (Level 2). Therefore, in conjunction with a relative abundance of agricultural values (Level 3), it does appear possible to estimate the total off-stream economic value of water employed along a supply chain (Figure 1). However, as for the broader question of relevant geographical variation in economic values (Figure 2) and the trade-offs that this might then enable, the relative abundance of agricultural values holds the most promise for developing a useful and generalisable method. Therefore, we now illustrate how geographical variations in economic value at Level 3 locations highlight the merit of a values-based approach and the trade-offs that it permits.

6.1. Case Study Illustration – Comparing Volume with Economic Value

Figure 4 presents a three-level durum wheat pasta supply chain map including the volumes of blue and grey water used to produce and consume one tonne of durum wheat pasta as a finished good.[13,14] Green water was excluded in this case study because of the difficulty of assigning an economic value to this water type as referred to earlier (Section 5.1). Durum wheat is first cultivated at Level 3 in Saskatchewan (Canada), Arizona (USA) and Baja California (Mexico). Factory-based milling and pasta processing then takes place in Italy (Level 2), before the pasta is consumed in Germany (Level 1).

Ruini *et al*. (2013) suggest that the ingredients used in the production of durum wheat pasta are semolina flour derived from durum wheat, and water.[15] The water burden associated with semolina is shown in Table 11. This is based on the volumes of water consumed and degraded in the cultivation of durum wheat

[13] We have used one tonne here rather than one kilogram (as is typical in water footprint studies) because larger production quantities are more meaningful units of analysis when the focus is economic values which only tend to register in cubic metres.
[14] This case study is loosely based on Ruini *et al*. (2013).
[15] Aldaya and Hoekstra (2010) suggest that salt is also present in the production of durum wheat pasta. However, they exclude salt from their analysis on the basis that it has an immaterial impact on the water footprint. Salt has therefore also been excluded in the analysis here.

at each Level 3 location, as estimated by Mekonnen and Hoekstra (2011). To allocate the water burdens associated with the primary crop (durum wheat) to crop-derived products (semolina), the approach used by Aldaya and Hoekstra (2010) involving the use of product and value fractions was adopted. It was assumed that 72% of the durum wheat is processed into semolina flour (the remainder is wheat bran and germ), and that semolina represents 88% of the total value of these two products. It is further assumed that an equal quantity of durum wheat (33.3%) is procured from each of the three locations at Level 3.

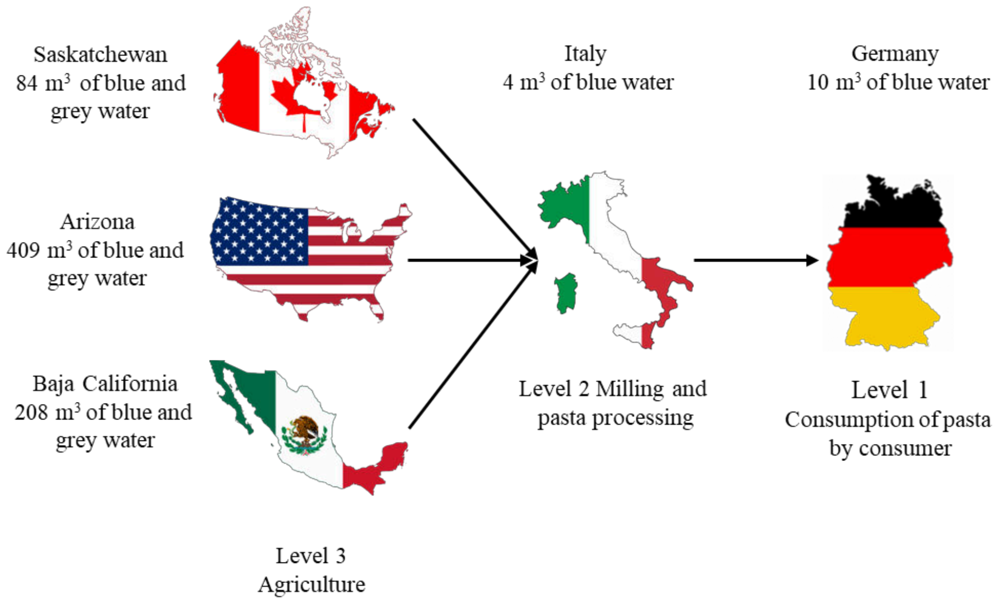

Figure 4. Durum wheat pasta supply chain map including volumes of blue and grey water used to produce one tonne of pasta as a finished good. Assumes: 1) the volumes of blue and grey water associated with durum wheat cultivation are allocated to semolina flour using a product fraction of 0.72 and a value fraction of 0.88, and 2) that an equal amount of durum wheat (33.3%) is sourced from each of the three locations at Level 3.

The volumes of water associated with the steps in pasta production at Level 2 – pre-cleaning and tidying up, conditioning, milling, raw material storage, mixing dough and rolling, drying, packaging, storage and distribution – all of which is blue water, has been taken from Ruini *et al.* (2013). In line with the approach adopted by Aldaya and Hoekstra (2010), it has been assumed here that the water used as an ingredient in pasta production is removed during the drying process. It was assumed by Ruini *et al.* (2013) that any wastewater associated pasta production at Level 2 is returned to a wastewater treatment plant and thus that there is no grey water footprint associated with this level in the supply chain. The volume of water at Level 1 has also been taken from Runin *et al*. (2013). Again, there is no grey water burden as it is assumed that any wastewater goes to a wastewater treatment plant.

Table 11
The water footprint of semolina flour processed from durum wheat cultivated at each Level 3 location

Location	Water footprint of raw material (Durum wheat) (m³/tonne) [a]		Product fraction [b]	Value fraction [b]	Water footprint of item (Semolina flour) (m³/tonne)		Total water footprint of item (Semolina flour) (m³/tonne)	33.3% of total water footprint of item (Semolina flour) (m³) [c]
	Blue water	Grey water			Blue water	Grey water		
Saskatchewan (Canada)	1.00	206.00	0.72	0.88	1.22	251.78	253.00	**84.25**
Arizona (USA)	848.00	156.00	0.72	0.88	1,036.44	190.67	1,227.11	**408.63**
Baja California (Mexico)	325.00	186.00	0.72	0.88	397.22	227.33	624.56	**207.98**

Note. USA = United States of America. m^3 = cubic metre. [a] Mekonnen and Hoekstra (2011). [b] Aldaya and Hoekstra (2010). [c] Assumes that an equal quantity of durum wheat is sourced from each location.

Table 12 sets out the economic value of the water at each supply chain level. The derivation of the unit economic values (i.e. per cubic metre) for each Level 3 location is shown in more detail in Appendix A. As presented in Appendix A, the unit economic values at Level 3 are averages across multiple primary estimates that have been recorded in Saskatchewan (3), Arizona (4) and Mexico (4).[16] These individual primary estimates have been selected from the 353 agricultural values (209 from the USA; 144 from the ROW) that were presented in Section 3. In the case of Saskatchewan and Arizona, there is a reasonable correspondence between study and policy sites. However, in Mexico, the unit value estimates originate from central Mexico rather than the north of the country where Baja California is located. We return to this point in our conclusions as this illustrates important limitations on what is achievable with the available data. All of the economic values in Appendix A are representative of water application (or unknown water volume measures) as none were available which reflected the value of water consumed i.e. the volumetric measure used to account for water volumes in the case study. As such, the values in Appendix A represent a lower bound estimate of the value of water in each location.[17]

The unit value at Level 2 is an average of those shown in Table 10 for water consumed in the food industry. The unit value at Level 1 has been estimated using Equation 1. The resulting at site value was

[16] The unit values have been chosen so that they are as similar as possible (i.e. at site, short-run values for applied irrigation water). Where possible we have used values for irrigation applied in wheat cultivation. If the authors did not estimate a value for wheat cultivation, we have instead used values for similar low valued crops.
[17] The value of water consumed in agriculture tends to be higher than that which is withdrawn or applied as it refers to the most productive part of irrigation i.e. the part that is usefully used by the crop during evapotranspiration.

calculated using a unit price estimate for domestic water supply in Berlin from Global Water Intelligence (2016) ($8.87), an estimate of price elasticity of 0.229 (Schleich and Hillenbrand, 2007), and an assumed 10% reduction in the quantity of water used from 115 litres per person per day (Environment Agency, 2008).[18]

Table 12

The volume and value of the blue and grey water employed to produce one tonne of durum wheat pasta

Level	Location	Blue and grey water (m^3) [a]	Unit value (2014 $/$m^3$)	Total value (2014 $)
3 (Durum wheat cultivation)	Saskatchewan (Canada)	84.25	0.10 [b]	8.42
3 (Durum wheat cultivation)	Arizona (USA)	408.63	0.08 [b]	32.69
3 (Durum wheat cultivation)	Baja California (Mexico)	207.98	0.24 [b]	49.91
2 (Milling and pasta processing)	Italy	4.00	2.39 [c]	9.56
1 (Consumption of pasta)	Germany (Berlin)	10.00	11.22	112.20
Levels 1-3		715		212.79

Note. USA = United States of America. m^3 = cubic metre. $ = United States Dollar. [a] See Table 11. [b] See Appendix A. [c] See Table 10.

In this scenario, the total volume of virtual water has a value ($212.79), which itself may be instructive when compared to other production inputs. However, it is relevant comparisons between functionally identical water use at each Level 3 location that highlight the merit of an economic approach and the trade-offs that it enables. Looked at through this lens, the optimum Level 3 sourcing location is the area with the lowest value of water, i.e. where the intensity of WTP was lowest or put another way, where the costs of water consumption and degradation are lowest. Indeed, when the focus is on different drops of water, as it is along a supply chain, then the traditional policy prescription from welfare economics (i.e. that water should flow to the highest valued user) is reversed. As shown in Table 12, the optimum Level 3 sourcing location for an economic value perspective, assuming that other input costs are constant, is clearly Saskatchewan ($8.42). This conclusion is in accordance with a volume perspective. However, the least attractive sourcing location from an economic value perspective is Baja California (highest total value of water) even though this location is responsible for a lower volume of blue and grey water when compared to Arizona (208 m^3 versus 409 m^3). Table 13 summarises the messages from this simple illustrative example that suggests introducing the value of water into alternative supply sourcing decisions might overturn decisions that are currently being made solely on volumetric grounds, i.e. the desire to minimise water use. At this point, a fuller explication of the method

[18] Price estimate is for combined water and wastewater and is based on a two-person household and monthly bill cycle (monthly usage falls into the > 6m^3 block tariff).

would also introduce sensitivity analysis to understand how sensitive these messages are to potential transfer errors.

Table 13
Optimum sourcing locations according to water volume and economic value perspectives

	Optimum Level 3 sourcing location based on volumes of blue and grey water	Optimum Level 3 sourcing location based on the economic value of blue and grey water
Preference 1	Saskatchewan (Canada)	Saskatchewan (Canada)
Preference 2	Baja California (Mexico)	Arizona (USA)
Preference 3	Arizona (USA)	Baja California (Mexico)

Note. USA = United States of America.

In addition to informing supply chain sourcing decisions, understanding water utilisation in economic terms may also encourage productive efficiencies at each supply chain level. For example, the disparity in unit values between Saskatchewan and Baja California may incentivise an increase in irrigation efficiency in Saskatchewan. Indeed, the concept of economic value applied here is a conceptually correct welfare measure. As such, it provides a better understanding of the real return to water in agriculture than the more simplistic notions of *economic water productivity* that currently seem to be favoured in the WFA literature, but which have no basis in microeconomic theory. [19]

Overall, for the agricultural level of a supply chain at least, the methodology outlined here can be used to estimate relative water impact in the form of variations in the intensity of WTP. As a result, it is relevant to LCA scholars who are interested in local water scarcity. However, by allocating in favour of areas with low WTP (i.e. economic value) and drawing attention to potential productive efficiencies that may be associated with this, the approach also focuses attention on water basins where water maybe abundant and being used inefficiently. Therefore, this methodology may offer an additional tool to WFA scholars – in addition to those already employed such as water footprint benchmarks and water footprint caps – who have focused on water abundant areas and the scope they offer to displace water in water scarce areas. Indeed, the principal message suggested by this research is that geographical disparities in economic value need to be considered so as to avoid incorrect sourcing decisions that may be suggested by looking at the volume of virtual water alone.

[19] For example, in the context of water used in agriculture, a conceptually correct welfare measure would seek to isolate the contribution that water makes after accounting for the contribution of other inputs. Economic water productivity, by contrast, does not consider the contribution of other inputs and thus greatly overstates the value measure.

7. Conclusion and Future Research

This paper has set out the assumptions and data requirements that would be necessary to place an economic value on all of the various water services encompassed by an agri-food supply chain at the product brand level. This paper has also demonstrated the extent to which it is possible to estimate these values with the data currently available in the literature.

The resulting method presented in Section 6 was not able to assign a value to in-stream ecosystem services and thus it was not able to approximate all the components of Total Economic Value. The method was also unable to assign a value to green water. However, the method can address off-stream water use at all levels of the supply chain. In particular, the relative abundance of economic values for off-stream water use in agriculture was able to illuminate the trade-offs that an economic value approach can suggest. Consequently, agricultural values are the focus of the method developed. Unlike conventional economic theory though, which advocates that water should flow to the highest valued use, this method indicates that water should be allocated to the lowest valued use (or location) within each functional context along a supply chain.

The questions we are left with here are twofold. First, to what extent is this method useful as it currently stands? Second, given the potential allocative efficiency gains indicated by the durum wheat pasta supply chain case study, to what extent would the method be useful if developed further?

In response to the first question, the method developed here can estimate functionally specific economic values for the water used at each level of an agri-food supply chain (i.e. Figure 1) and thus the total value of off-stream virtual water. However, the ability of the method to indicate geographical variation in economic value within the agricultural level of a supply chain (Figure 2) requires careful selection from what is a limited range of value estimates. For example, in the durum wheat pasta case study, there was not a robust value for the specific region in Mexico, so a value from a different region in the country had to be used instead. Indeed, until more data are available, this proposed methodology should be considered a limited tool. For example, it might be useful in the initial screening of supply chains.

However, with regard to whether this proposed methodology could be useful if developed further, the durum wheat pasta case study has shown how assigning a value to virtual water provides a clear and easily comprehensible summary of the impact of water use, as well as the trade-offs associated with this. Therefore, we would suggest that if the method outlined here were to be developed further, it could be more intuitively appealing to supply chain managers than the complex stress-weighted water footprint approach. Moreover, the method could also be relevant to the water footprint community and those who want to focus on displacing water use in water scarce areas by allocating production to areas of water abundance.

To develop the method further, this paper highlights several areas in the environmental valuation and management literature that need advancing. Foremost amongst these, there were only a limited number of papers available, many of which are now somewhat dated. Indeed, some water categories, specifically waste assimilation, appear to have gone out of fashion altogether with no significant contributions occurring since the 1970s. Therefore, more original studies deriving primary data are needed to update and add to the literature available on the value of water. In particular, where possible estimates should be denominated in volumetric terms as these would appear to have the greatest potential relevance for business and supply chain managers. In addition, the value of water as an intermediate input into production (in particular in industry), the value of green water, the unit valuation of in-stream ecosystem services, and a set of consistent standards for the reporting of valuation studies are all areas that could be augmented and refined. Doing so would further facilitate an approach that suggests the need for geographical disparities in economic value to inform sustainable sourcing and supply chain management decisions, and not just water volumes alone.

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Appendix A

Agricultural/irrigation water unit values used at Level 3 in pasta supply chain case study

Supply chain location at Level 3 (Policy site)	Source	Value type	At site/ at source	Long run/short-run	Water volume measure	Crop value	2014 USD per m^3	Study location (Study site)
Canada (Saskatchewan)	Bruneau (2007)	AV	At site	Unknown	Application	Low (wheat)	0.16	Saskatchewan and Alberta
Canada (Saskatchewan)	Samarawickrema & Kulshreshtha (2008)	AV	Unknown	Short	Application	Various – mostly low value	0.05	Two basins in Saskatchewan [a]
Canada (Saskatchewan)	Kulshreshtha and Brown (1990)	AV	Unknown	Short	Unknown	Various – mostly low value	0.09	Saskatchewan
AVERAGE							**0.10**	
USA (Arizona)	Bush & Martin (1986)	MV	At site	Short	Application	Low (wheat)	**0.10**	Arizona – Maricopa County
USA (Arizona)	Bush & Martin (1986)	MV	At site	Short	Application	Low (wheat)	**0.09**	Arizona – Pima County
USA (Arizona)	Bush & Martin (1986)	MV	At site	Short	Application	Low (wheat)	**0.09**	Arizona – (Pinal County
USA (Arizona)	Gibbons (1986)	MV	At site	Unknown	Application	Low (wheat)	**0.04**	Arizona
AVERAGE							**0.08**	
Mexico (Baja California)	Puente Gonzalez (2007) *in* EVRI (No date)	Unknown	Unknown	Unknown	Unknown	Low (Maize)	0.15	Veracruz
Mexico (Baja California)	Arias Rojo (2007) *in* EVRI (No date)	Unknown	Unknown	Unknown	Unknown	Unclear	0.32	Saltillo
Mexico (Baja California)	Zetina Espinosa *et al.* (2013)	MV	At site	Unknown	Unknown	Various – mostly low value	0.5 [b]	Hidalgo
Mexico (Baja California)	Zetina Espinosa *et al.* (2013)	MV	At site	Unknown	Unknown	Various – mostly low value	0.02 [c]	Hidalgo
AVERAGE							**0.24**	

MV = Marginal Value. AV = Average Value. [a] Unit value is an average across two sub-basins within Saskatchewan that are part of the South Saskatchewan River basin. [b] Median value in range given for winter season. [c] Median value in range given for summer season. Values converted from local currency to 2014 USD using World bank PPP exchange rates for GDP and the Implicit Price deflator for GDP from the Bureau of Economic Analysis (World Bank, 2016; BEA, 2016).