Monetary valuation of virtual water use in global supply chains

Benjamin H Lowe

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Abstract

The aim of this thesis has been to develop a new method that can be used to place a monetary figure, reflecting full economic and societal value, on the volumes of fresh water that are consumed and degraded in agri-food product supply chains. Informed by the twin concepts of Total Economic Value and Ecosystem Services, a detailed review of the water valuation literature, which had been conducted within a welfare economic framework, suggested that the current evidence base is limited in terms of the number, type, coverage and robustness of existing estimates. Nonetheless, a method is developed which looks to provide an estimate of the *direct use value* of water in three agri-food supply chain case studies which are underpinned by raw materials that either significantly impact, or impacted by, global freshwater resources (wheat, tea and potatoes). These case studies are used to illustrate the merit of such an approach in terms of assessing the relative scarcity or impact of water use along globally disparate supply chains, and as a means promoting the trade-offs associated with productive and allocative efficiency gains. Indeed, it is argued that the principal contribution of the thesis is that it highlights the potential for the academic community to enable a more comprehensive approach to the valuation of virtual water flows. Such an approach would supplement the volumetric focus of water footprint assessment, and provide a more useful metric for business users than the current focus on the stress weighted water footprint.

Key words:

Virtual water, Total Economic Value, Ecosystem Services, water footprint, stress weighted water footprint.

Outputs and qualifications arising from this thesis

Papers

Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2016). A proposed new method for placing monetary values on virtual water to improve the efficiency of global supply chains. Paper presented at the British Academy of Management Annual Conference 2016, *Thriving in Turbulent* Times, 6th– 8th September.

Posters

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Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2016). *Monetising the product water footprint for sustainability assessment*. [Poster]. White Rose Doctoral Training Centre (Business pathway) *Change*, 7th – 8th July, The University of Leeds.

Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2015). *Economic valuation of virtual water across supply chains*. [Poster]. White Rose Doctoral Training Centre (Business pathway) *Sustainability*, 11th – 12th June, The University of Sheffield.

Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2015). *Economic valuation of virtual water across supply chains*. [Poster]. White Rose Doctoral Training Centre (Planning pathway) *Planning for Impact*, 19th May, The University of Sheffield.

Reports

Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2016). The potato crisp water footprint Stage 1 report. Report prepared for [anonymised company]. The University of Sheffield.

Lowe, B.H., Oglethorpe, D. and Choudhary, S. (2016). The potato crisp water footprint Stage 2 report. Report prepared for [anonymised company]. The University of Sheffield.

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Abbreviations

BOD	Biological Oxygen Demand
BT	Benefit Transfer
CDP	Carbon Disclosure Project
CSR	Corporate Social Responsibility
CWU	Crop Water Use
ESS	Ecosystem Services
ET	Evapotranspiration
EU	European Union
EVRI	Environmental Valuation Reference Inventory
LCA	Life Cycle Analysis
MENA	Middle East and North Africa
MPC	Marginal Private Cost
MSC	Marginal Social Cost
OECD	Organisation for Economic Co-operation and Development
PGP	Provider Gets Principle
PPP	Polluter Pays Principle
RP	Revealed Preference
RQ	Research Question
SKU	Stock Keeping Unit
SP	Stated Preference
TEV	Total Economic Value
WFA	Water Footprint Assessment
WFN	Water Footprint Network
WSI	Water Stress Index
WTA	Willingness to Accept
WTP	Willingness to Pay
Y	Yield

Units and conversions

1 kilogram	1,000 grams
1 tonne	1,000 kilograms
1 acre foot	1,233.48 cubic metres
1 cubic metre	1,000 litres
1 litre	1 kilogram
1 cubic metre	1 tonne

1. Introduction

1.1 Background and motivation

Each year the World Economic Forum produces a Global Risks Report. In the 2015 edition, 'looming' freshwater crises were considered to be the most significant long-term global risk in terms of potential impact, across all of the economic, environmental, geopolitical, societal and technological risk categories that were assessed.[1,2] The prominence of water crises as a pressing global risk was reaffirmed in 2016, and again in 2017, when water crises were ranked third for potential impact behind weapons of mass destruction (both years), failure to adapt to climate change (2016) and extreme weather events (2017). Whilst this may seem unlikely given that approximately 70% of the earth's surface is covered by water, crucially less than 1% of all the earth's water resources are *easily accessible* freshwater, and even though this is a renewable resource, it is subject to profound spatial and temporal disparities. Moreover, freshwater resources are also threatened by multiple and interrelated socio-economic, demographic and environmental pressures, all of which are becoming increasingly insistent, and which, in concert, suggest that by 2030 global water requirements will exceed sustainable supplies by 40% (2030 Water Resources Group, 2009).

Agri-food businesses are in the front line of this crisis as their operations are both sustained by water resources, but also significantly contribute to water scarcity given that, globally, approximately 70-80% of all water is consumed in agriculture. Indeed, CERES (2017) suggest that so far this year, 90 major food sector companies have highlighted water risks in their earnings calls, and 85% of all the companies in CERES's annual food company tracking have reported water as a material risk in their financial returns.

Against this backdrop, the concept of virtual water – the volume of water used along a supply chain to produce the products and services we consume – has gained substantial traction as a means of understanding how the production and consumption of products in one location often impacts watersheds in other, globally disparate, locations. Indeed,

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[1] The World Economic Forum defines global risks as 'an uncertain event or condition that, if it occurs, can cause significant negative impact for several countries or industries within the next ten years.' Water crises are defined as 'a significant decline in the available quality and quantity of fresh water, resulting in harmful effects on human health and/or economic activity' (World Economic Forum, 2015).
[2] Throughout this thesis 'water' and 'freshwater' will be considered synonymous. All other categories of water (e.g. salt water) will be referred to as such.

the virtual water concept has been used to show that, particularly with agri-food products, it is this ‘hidden’ component of water dependency associated with indirect water use in the supply chain, rather than that used in direct operations, that often represents by far the largest appropriation of freshwater. For example, Ercin *et al.* (2011) suggest that 99% of all the water that is consumed in the production of a carbonated beverage is associated with the supply chain ingredients, particularly sugar. In spite of this, as the surveys conducted by CERES (2015) and The Carbon Disclosure Project (CDP) (2014) show, this indirect use of water in the supply chain is still somewhat of a business blind spot even though it is the complex and geographically diverse nature of the supply chain itself that often ensures that it is the first to suffer in the face of water related events. Indeed, this has led to a drive to improve the reporting of water use by businesses in the form of the Alliance for Water Stewardship Standard and the Carbon Disclosure Project Global Water Report, amongst others.

However, the very appeal of the virtual water concept and its later evolution into the water footprint (Hoekstra *et al.* 2011), as well as the drive for better water reporting, all arise because the value of water is largely not already influencing its efficient allocation. Indeed, markets and the signals that they provide about relative scarcity, have a very limited role in this process because of a number of market and institutional failures that are associated with the particular characteristics of water and the various uses that it is put to. At the macro level, the most telling result of this has been the ensuing perverse market incentives which have seen water intensive products produced in areas where water is scarce, and exported to areas of relative water abundance. For example, in India, the northern states which experience significant water stress have been exporting large volumes of water intensive food produce to states in the east of the country which have far greater water endowments (Verma *et al*. 2009). At a more local level, water is often allocated to sectors where it has a low inherent value at the expense of sectors where it can be put to higher valued ends.

The merit of valuing water correctly was recognised, most notably, in the Dublin Declaration in 1992. Principle Four of the declaration states the following:

> *‘Water has an economic value in all its competing uses and should be recognised as an economic good.’ ‘Within this principle, it is vital to recognise first the basic right of all human beings to have access to clean water and*

> *sanitation at an affordable price.* *<u>Past failure to recognize the economic value of water has led to wasteful and environmentally damaging uses of the resource. Managing water as an economic good is an important way of achieving efficient and equitable use, and of encouraging conservation and protection of water resources'</u>* (emphasis added) (The Dublin Declaration, 1992).

Within mainstream or welfare economics the area of environmental valuation provides a number of methods which look to estimate shadow prices for environmental goods and services, including water, where market prices are either entirely absent or a poor indicator of value. These methods are a means of ultimately promoting more efficient resource allocation and management by evaluating the trade-offs, in monetary terms, associated with competing uses, also taking into account dis-benefits (economic costs) such as pollution, often within a cost-benefit analysis framework. These methods have been accepted and implemented by governments and organisations around the world, including the World Bank, the Environmental Protection Agency and the National Oceanic and Atmospheric Administration (both in the USA), and numerous government departments in the UK. For example, environmental valuation has been used by the Environment Agency as part of the drafting of the Marine and Coastal Access Act 2009 which established marine conservation zones around the UK, and water company business plans in England and Wales are based on an assessment of their impact on water quality and ecosystem services (Morrison and MacDonald, 2010, p.14). In addition, the methods have been used in high profile cases such as the *Exxon Valdez* oil tanker spillage that occurred in Alaska in 1989, and more recently, the *Deepwater Horizon* spill in 2010. Here environmental valuation techniques were used to assess the damages to environmental resources for the purposes of litigation, and ultimately, the internalisation of a negative externality.

To date, however, the domains of environmental valuation and virtual water have not interacted to any great degree in the academic literature. This is despite the fact that there are increasing moves by businesses to incorporate the value of natural capital stocks and flows into decision making. Of particular relevance here is the Natural Capital Protocol which was adopted in 2016 by the Natural Capital Coalition which is made up of 250 leading businesses. This provides a standardised *framework* for the valuation and assessment of impacts and dependencies on a wide variety of natural capital stocks and flows along the supply chain. As we will see though, whilst the economic valuation of virtual water has received some attention in the grey literature as a means of assessing the impact and risk associated with water use across geographically diverse supply chains that encompass differing levels of water scarcity, in a metric which businesses understand and one which permits a direct comparison with other inputs in production, it remains a virtually unstudied area in academia. As a result, the principal working aim of this thesis is set out below:

To assess the feasibility of, and means to achieve, the measurement of the economic and societal value of virtual water, expressed in monetary terms, within selected global supply chain case studies. Moreover, to explore how this may improve the efficiency of intra-supply chain water usage.

As will be introduced at greater length in what follows, in this context economic value refers to financial or private values which are based on actual financial transactions. Societal values, also known as public values, are typically not accounted for by companies, and are thus labelled externalities or third-party impacts. Numerous environmental and social values fall into the category of societal values given that they are not subject to the market mechanism and thus have no market price (WBCSD, 2013, p.22). Full value can be thought of here as economic and societal value.

As the literature review in Chapter Two will reveal, four principal research questions are developed in order to address the overall working aim:

1. Can the existing body of environmental valuation literature support the estimation of unit values of fresh water use that can be transferred to the multiple geographies that global supply chains encompass?
2. How is the full value of virtual water, within selected supply chain case studies, distributed by: 1) supply chain stage, and 2) geography?
3. What does the inclusion of a measure of the full value of virtual water reveal about the efficient use and allocation of water in supply chains?
4. How can regulatory instruments be designed in response to the full value of virtual water and its relative distribution in supply chains?

1.2 Thesis structure

The thesis is structured as follows. Chapter Two reviews the body of literature relevant to the thesis, draws out the knowledge gaps identified, and develops the corresponding research questions that have been selected for pursuit. Chapter Three outlines the methodology that has been designed to address these questions, which is then applied in each of three case studies that have been selected for analysis. Chapter Four presents the first of these case studies which is based on the durum wheat pasta supply chain, followed in Chapter Five by the second case study which is based on the tea supply chain. Both the pasta and tea case studies are exclusively based on secondary data, in contrast to the case study presented in Chapter Six which sets out the potato crisp supply chain, volumetric water data for which, has been obtained from the primary source detailed. As will be elaborated in what follows, each of the case studies has been selected because agriculture is the largest user of water globally and the raw materials associated with each supply chain (wheat, tea and potatoes) either significantly impact, or are impacted by, freshwater resources.

Chapter Seven addresses the conclusions that stem directly from the research, as well as the policy implications and a number of recommendations that the environmental valuation discipline might adopt in the future. Finally, Chapter Eight synthesises the results with the theoretical and methodological context, as well as detailing the authors self-reflections and considerations of a future research agenda.

2. Literature review

Parts of this chapter were presented at the 2016 British Academy of Management Conference in a paper titled "A *proposed new method for placing monetary values on virtual water to improve the efficiency of global supply chains*."

The opening question posed on page four focuses the scope of this review on the bodies of literature detailed below, each of which will be covered in the following sub-sections.

1. Theoretical insights into the valuation of water resources and what, in terms of welfare economics, constitutes efficient resource utilisation (Section 2.1).
2. Research on economic policy instruments, derived from welfare economic theory, and their role in affecting such utilisation (Section 2.2).
3. Research on the empirical measurement and assessment of virtual water flows (Section 2.3).

Following this, section 2.4 will outline the research questions that the literature review has identified and which will be addressed in the remainder of the thesis.

2.1 Economic valuation of water resources

This section begins with a brief discussion of different philosophical conceptions of what constitutes 'value' before introducing the principal tenets of welfare economic theory, the valuation techniques that emanate from these, and the literature which has looked to apply them in a water context.

2.1.1 What do we mean by value?

There is a large amount of philosophical, not to mention cultural and even ethical, complexity underpinning the answer to this question, the vast majority of which is beyond the scope of this thesis (Hines 1991; Fourcade 2011; Gomez-Baggethun and Ruiz-Perez 2011; Sullivan 2014). However, as a starting point, a useful distinction is made by philosophers between the notions of intrinsic and extrinsic value, which can be broadly presented as follows:

> *'That which is intrinsically good is nonderivatively good; it is good for its own sake. That which is not intrinsically good but extrinsically good is derivatively good; it is good not…for its own sake, but for the sake of something else that is good and to which it is related in some way'* (Zimmerman, 2004 *in* Ozdemiroglu *et al.* 2006, p. 6/7).

The majority of welfare economics, in keeping with its foundation in the utilitarian traditions of Bentham and Mill, assumes that ‘pleasure is intrinsically good (and pain intrinsically bad), generally narrowing this to an anthropocentric (human centred) focus on pleasure and pain’ (Ozdemiroglu *et al.* 2006, p.7). Furthermore, this tradition also assumes that individual preferences are a reliable indicator of the relative pleasure of different outcomes (*Ibid*). As a result, the economic value of natural resources arises because they provide environmental goods and services which ‘satisfy human needs and wants [and thus] increase the well-being or utility of individuals’ (Champ *et al.* 2003, p.9).

Table 2.1 applies these philosophical distinctions to the natural environment. The majority of value assessments, following the utilitarian tradition, stem from the two shaded boxes (i.e. anthropocentric values), which will also be the focus of this thesis. Whilst recognising the argument that other living things may have their own values, and that the environment has value in and of itself, given the working question posed, the scope of what follows will be limited to values that are measurable (i.e. anthropocentric values) and the techniques used to estimate them.

Table 2.1 Classification of environmental value

	Anthropocentric	**Non-anthropocentric**
Instrumental	Total Economic Value: personal use and non-use (including existence value related to others’ use).	The values of other animals, species, ecosystems etc. (independent of humans).
Intrinsic	‘Stewardship’ value (unrelated to any human use).	Value an entity possesses independently of any valuer.

Source: Ozdemiroglu *et al.*, 2006, p. 7.

2.1.2 Valuation and welfare

In welfare economics, economic value stems from the impact that a good or resource has on social welfare, which in turn, is derived from the aggregate impact on the utility of individuals in society. The utility of individuals is determined by their preferences which are conveyed by the amount that they are willing to pay (WTP) for goods and services, specified in monetary terms (Turner *et al.*, 2004, p. 53). The resulting market prices then, in the absence of market distortions, represent economic values which provide the foundation for evaluating the trade-offs associated with the allocation of resources between alternative and competing wants. In competitive market situation, allocative efficiency is achieved when demand is equal to supply, and with it, when marginal cost is equal to marginal benefit as shown in Figure 2.1 below at point *Pm* and *Qm*. This situation maximises social welfare and achieves a Pareto optimal outcome where no reallocation can be attempted which would increase the welfare of one individual without making another worse off. Moreover, in equilibrium, the value of a good or resource is maximised across all economic sectors, with allocation in favour of high value uses at the expense of low value uses.

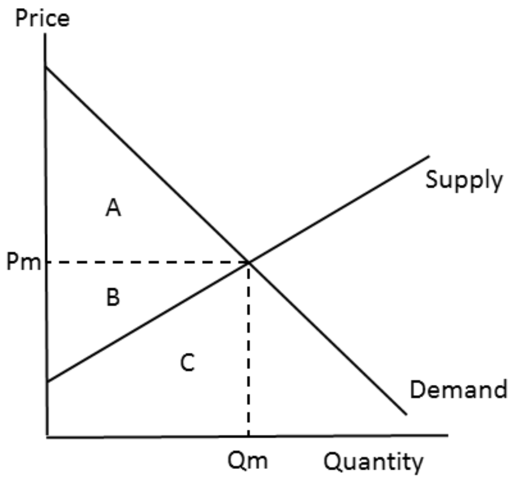

Figure 2.1 Allocative efficiency in equilibrium (source: author).

2.1.3 Reasons for inefficient allocation - the demand for environmental values

The principal driving force behind the need to value environmental goods and services, generally, stems from the fact that many of these goods and services are intangible and thus not traded on markets. As a result, there are no price signals available through which society can indicate their preferences which may lead to over exploitation. Examples of goods and services which are intangible, in a water context, include the role that water can play in assimilating waste, providing habitat for wildlife, and giving rise to recreational experiences. Indeed, many of these goods and services are examples of public or common pool goods which have characteristics of non-rivalry and non-excludability and thus are an example of market failure and therefore sub-optimal resource allocation. However, whether the good in question is public (such as those just mentioned) or private (such as water that is used in agriculture and industry), two additional market failures are also closely associated with water resources: first, the presence of externalities, and second, open access pressures which stem from the inability to establish property rights and thus gives rise to the ‘tragedy of the commons.’ The classic example of a negative externality in the form of water pollution, which is pertinent in this context, is illustrated in Figure 2.2 below. Here, there is a divergence between marginal private cost (MPC) of the activity and marginal social costs (MSC) of the activity equal to the externality. Consequently, the market over produces (Q1 as opposed to Q*) and the price charged is too low (P1 as opposed to P*).

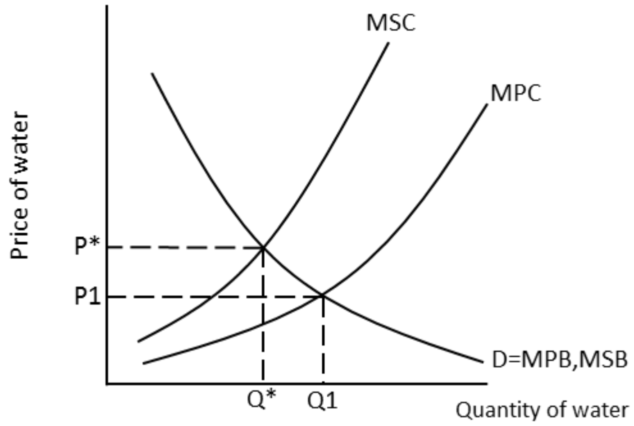

Figure 2.2 Sub-optimal resource allocation associated with negative externalities (source: author).

In addition to the three 'classic' market failures listed above which are applicable to natural resources in general, Savenije (2002), Hanemann (2006) and Young and Loomis (2014) all set out the particular characteristics of water resources that further ensure the absence or ineffectiveness of markets and thus that impede the formation of a market price which is a true indication of its value. In brief, these include: 1) raw water supplies are unpredictable in time, space and quality; 2) significant economies of scale exist, especially in municipal supply, which lend themselves to public price regulation in order to avoid monopolistic pricing; 3) capital and energy costs associated with the transportation, extraction, and storage of water tend to be high relative to economic value at the point of use (Young and Loomis, 2014, p.6), and 4) given the essential nature of water for life and sanitation, many suggest that water regulatory approaches are more appropriate than market mechanisms. Indeed, as Hanemann (2006, p.76) argues, the end result of many of these impediments to the operation of markets is that the 'prices which most users pay for water reflect, at best, its physical supply cost and not its scarcity value.'

2.1.4 How do we value natural resources? A conceptual framework

Within the framework of environmental valuation, which stems from mainstream welfare economics, the valuation of natural resources draws on the twin concepts of

WTP (for an enhancement in environmental provision or to avoid a decline) and willingness to accept (WTA) (to sacrifice an enhancement in environmental provision or accept a decline). These concepts are expressions of preferences, but their primary purpose is the quantification of the variations in individual, and thus societal, welfare, that are caused by changes in environmental goods and services. As such, WTP and WTA are linked to three specific welfare measures in microeconomic theory: *consumer surplus* as measured using a traditional Marshallian demand curve, and the more precise *compensating* and *equivalent* measures which are derived from Hicksian demand curves. In this context, the consumer surplus measure, as applied to a marketed commodity, is used for illustration purposes as it provides an accessible understanding of the concepts involved. However, for a fuller overview of the Marshallian and Hicksian measures, and their application to marketed and non-marketed commodities, see Champ *et al.* (2003).It should be noted that the consumer surplus suffers from the fact that it keeps the marginal utility of income constant and therefore is most appropriate for goods where the quantity demanded is not dependent on income. Nonetheless, as Young and Loomis (2014, p.32) argue, where the good in question only accounts for a small portion of the household budget, as is the case with the majority of services provided by water, the Marshallian measure of consumer surplus is a close approximation of the two Hicksian measures and therefore suitable for most practical applications (Young and Loomis, 2014, p.32).

Figure 2.1 above depicts the consumer surplus as the area below the demand curve but above the price line (area A). The consumer surplus represents the ‘difference between the maximum that users would be willing to pay and what they would actually have to pay under a constant price per unit (Young and Loomis, 2014, p.32). Put another way, if quantities less than *Qm* are traded, consumers’ WTP, as represented by the demand curve, is in excess of market price. Those consumers who are willing to pay more than *Pm* are gaining additional utility over and above the price paid, equal to area A. This suggests that market prices and economic value are not synonymous and that the former is only a lower range estimate of the latter. Total social benefits are represented by area B, the *producer surplus* (which arises because producers will sell for less than the market price if the quantity traded is less than *Qm*), and C, the cost of producing *Qm*, plus the consumer surplus (Turner *et al*., 2004, p. 50). However, net social benefits are given by the sum of the consumer and producer surpluses. Three types of values can be identified in Figure 2.1: 1) total values which are represented by areas A, B and C divided by *Qm*,

2) average value which is the consumer surplus (area A) divided by *Qm*, and 3) marginal values which is the value of the last unit or *Qm* (see Kulshreshtha, 1994, p.20). Marginal values are of most use when it comes to decisions on the allocation of resources.

2.1.5 A taxonomy of water values

A taxonomy of water values is developed below which highlights the attributes of water resources that influence the nature of the economic values that are estimated. Beginning with the Total Economic Value and Ecosystem Services concepts – which outline the full range of benefits that natural resources provide – the value 'denominations' that can be estimated, together with the physical and economic dimensions that influence the value of water, are presented.

Total Economic Value and Ecosystem Services concepts

Given that the market failures referred to above often ensure that market price, if indeed the water use in question has one, does not adequately reflect benefit in use or the range of benefits that accrue from using the resource, there are several conceptual frameworks for classifying the full range of values which are linked to the goods and services provided by natural resources, including water (e.g. Turner and Postle 1994; Young 1996; Rogers *et al.* 1998). However, following Pearce and Turner (1990), the most widespread approach delineates the (additive) components of Total Economic Value (TEV) as shown in Figure 2.3 and explained in Table 2.2 below.

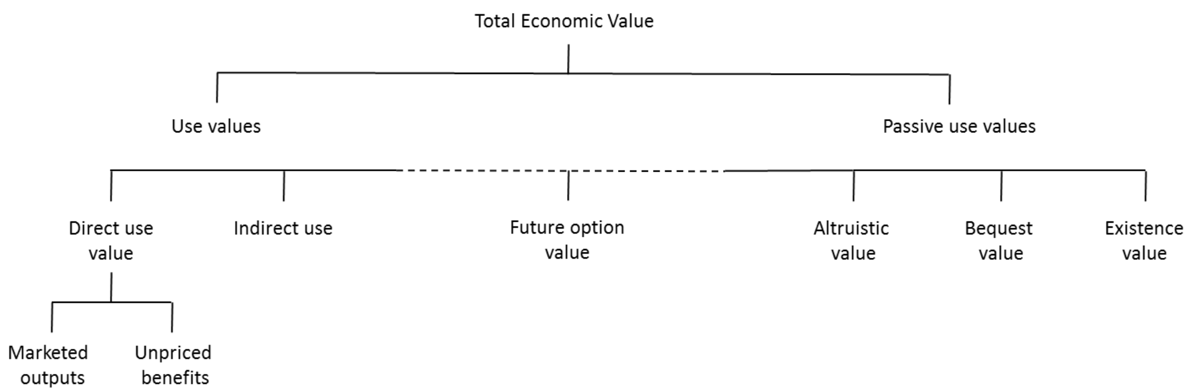

Figure 2.3 Total Economic Value framework (source: adapted from Marcouiller and Coggins, 1999 and Morrison and McDonald, 2010).

Table 2.2 Components of TEV

Use value	Relates to current or future uses of a good or services.
Direct use	Direct use value may be 'marketed outputs' (e.g. timber) or 'unpriced benefits' (e.g. recreation).
Indirect use	Indirect use values include key Ecosystem Services (e.g. climate regulation, flood protection, etc.).
Option value	Associated with retaining the option to use a resource in the future.
Non-use/passive use values	Derived from the knowledge that environmental resources continue to exist (existence value), or are available for others to use now (altruistic value) or in the future (bequest value).

Source: adapted from Bateman *et al.*, 2009, p. 3.

As can be seen, TEV is an anthropocentric framework which reflects how humans interact with the full range of goods and services provided by the natural environment. As such, it suggests how these goods and services impinge on societal welfare and thus provides a measure of full societal benefit. In splitting values into their use and non-use (also referred to as passive use) components, TEV includes both instrumental (use) and intrinsic (non-use) values. In addition, it reflects those direct use values that can be expressed in financial terms based on data from actual markets (economic or private values), as well as those use and non-use values which are not subject to the market mechanism (i.e. are non-marketed societal or public values). Moving forward in this thesis, the idea of 'full value' will be considered synonymous with TEV and the idea that it is including both economic and societal values. Specifically, in the context of water resources, use values include the value of water when used as an intermediate input in production, for example irrigation water used in agriculture or the water used in industry to produce goods and services. Non-use values, on the other hand, do not arise when water is used directly, but rather, from the knowledge that water resources exist (existence value), and are available for current (altruistic value) and future (bequest value) generations. Non-use value are thus public goods in that they are non-rival and non-excludable. In between use and non-use value, indirect use value refers to the hydrological services provided by water (such as flood control, sediment retention and ground water recharge), as well, for example, the benefits that water resources give rise to in the form of wildlife habitat. Indeed, the distinction between direct and indirect use values speaks to another classification that is important here: that between off-stream (extractive) and in-stream (at source) uses. The former refers to situations when water is removed from a stream for use in agricultural, industrial and municipal settings. The latter refers to the value of water *in situ* in providing hydrological, wildlife habitat and recreational benefits, amongst others.

On a point of detail, the notion of option value is the subject of debate in the literature with some taxonomies including it under use values and others within non-use values, hence the dotted line in Figure 2.3 More importantly, however option value is categorised, there are questions regarding whether it should be estimated separately given that it is viewed as more of a theoretical curiosity (Freeman 1993; Morrison and MacDonald 2010).

The Ecosystem Services (ESS) approach (see Costanza *et al.*, 1997; Millennium Ecosystem Assessment, 2005; Haines-Young and Potschin, 2013) is a means of categorising and understanding 'the linkages in the ecosystems that ultimately contribute to human welfare both through the provision of goods and services (use value) and non-use value' (Ozdemiroglu *et al.* 2006, p.10). Whilst it is separate from the TEV framework, nonetheless as shown in Figure 2.4 below, different water related ESS correspond to different components of TEV. This correspondence between the two

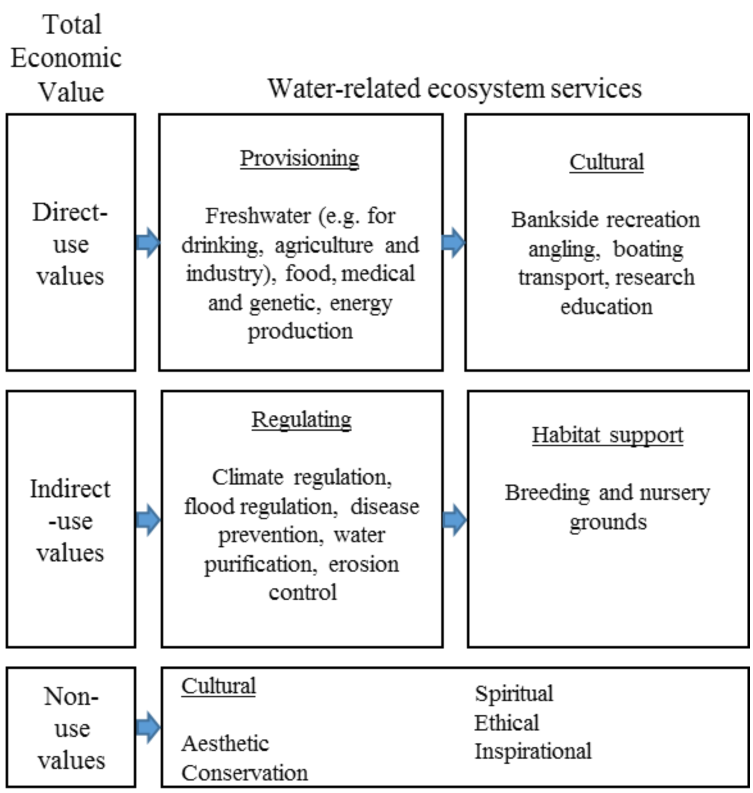

Figure 2.4 Ecosystem Services and the TEV framework (source: adapted from WBCSD, 2013, p. 26).

frameworks is something that will be drawn on in what follows as many economic values are estimated in terms of their functional uses (i.e. ESS) rather than the components of TEV, but by estimating the value of the ESS impacted by water resources an approximation of TEV can be achieved.

Value denominations

Water values are available in different 'denominations.' Whereas the focus here is on what will be referred to as *unit values* (i.e. values per cubic metre or acre foot) given that the subject of interest is the value of a certain *volume* of virtual water use, water values are also available as values per acre of land area (e.g. for irrigation water) as well as values per activity day (principally for recreational uses of water such as fishing), amongst others. Indeed, as will be referred to in what follows, a key aspect of this thesis will be the gathering and analysis of the unit values of water that are available and which correspond to the different components of the TEV framework.

Physical attributes

Where water values are available in volumetric terms, several different concepts of what constitutes a unit of water are available:

- Water *withdrawal* - the volume of water that is withdrawn from a surface or groundwater source.
- Water *delivery* or *application* – the quantity of water that is delivered to the location where it will be used. Water delivery will be less than water withdrawal depending upon how much water is lost in the process of moving it between the place of withdrawal and the place of application.
- Water *consumption* – refers to 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.), or because it has been incorporated into a crop or product.

Economic dimensions

There are a number of different economic dimensions of water (see Young and Loomis, 2014). First, the accounting stance can be either *private* or *social*. The former 'measures impacts in terms of the prices faced by the economic actors being studied,' whereas in the latter, 'social prices are those adjusted for taxes, subsidies, and other interventions'

(*Ibid*, p.35). Second, water values may be *short run* or *long run*. This distinction is predominantly applicable where water is an intermediate input into production (i.e. in agriculture and industry) and refers to whether or not fixed costs are taken in to account when deriving water values. For example, in the residual value method which is referred to further below and will be invoked at numerous points in what follows, the value attributable to irrigation water is derived from total revenue received for the crop, less all non-water input costs. Where these costs include fixed costs, the value can be said to be a long run value. Conversely, where they do not, the value is said to be short run. Third, water values can be *at site* (off stream) or *at source* (in stream). This distinction arises depending on whether or not any costs incurred in extracting the water from the stream and making use of it are included when deriving the water value. Using the residual value method as an example again, where these costs are not deducted, the value is said to be at site, where they are deducted, the value is said to be in stream. Finally, water values can be derived for a single period or instance (*per period*) or a stream of future values can be used to estimate a present capitalised value (*capital asset* value).

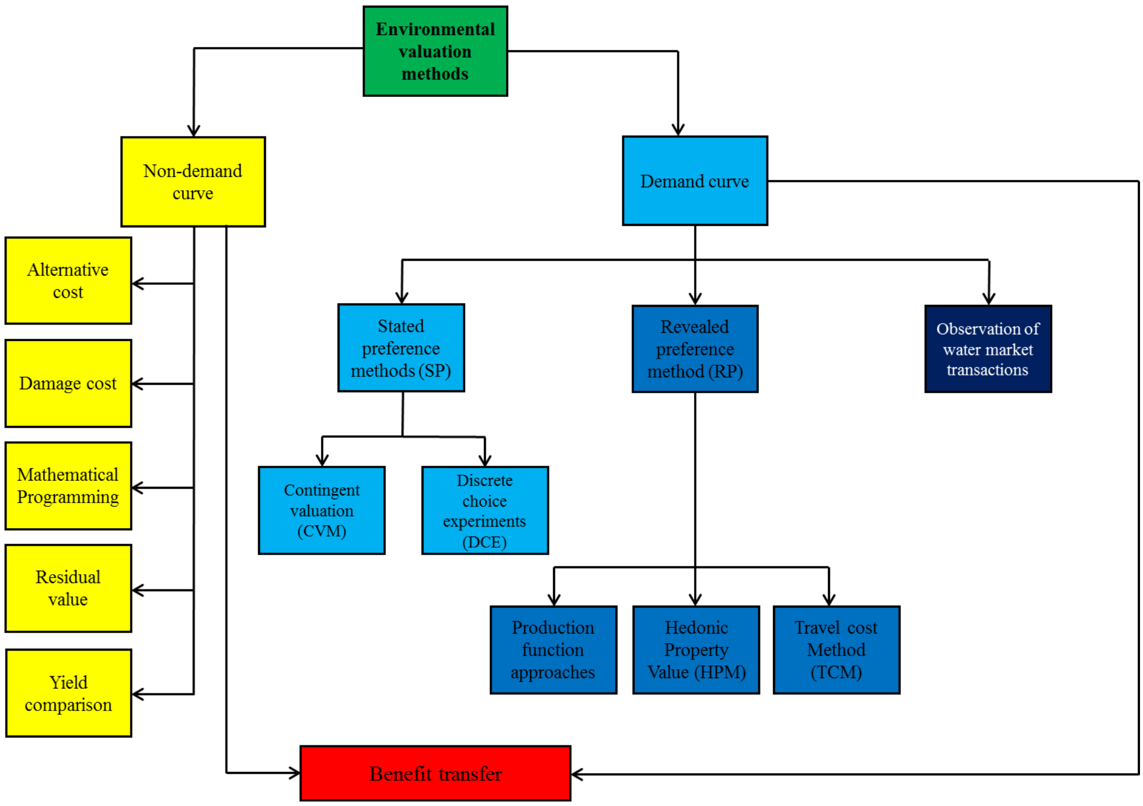

Figure 2.5 Principal Economic valuation methods (source: author).

2.1.6 Economic valuation techniques

Environmental valuation techniques provide different means of attempting to estimate WTP or WTA when this is not available in the form of market prices. There are multiple approaches to classifying economic valuation methods in the literature (see Hanley *et al.* 2007; Young and Loomis, 2014). However, a useful distinction that is made is between those methods which rely on the demand curve – and thus yield true welfare measures either in terms of Marshallian consumer surplus (see Figure 2.1) or Hicksian compensated demand curves – and those that do not. As shown in Figure 2.5 above, demand curve approaches can be further sub-divided into stated and revealed preference methods, as well as, where a market is available, the direct observation of market transactions. Where a market is not available, revealed preference (RP) methods draw on information from related markets to attribute values. A related market is 'one that indirectly reveals values for a good; that is, there is some relationship between prices paid in a market and environmental characteristics of a good, allowing value to be imputed' (Morrison and MacDonald, 2010, p.15). Stated preference (SP) methods utilise 'surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods' (Bateman *et al.*, 2011, p.1073).

Table 2.3 below provides a brief overview of the principal techniques that are used to value water resources, together with the component(s) of TEV that they are able to capture.

Table 2.3 Principal economic valuation techniques associated with freshwater resources

Valuation method	Demand curve?	Elements of TEV	Description of method and data source	Useful for valuing water as:
Alternative cost	No	Direct & indirect use	Value attributable to cost savings from next best alternative source of service (e.g. water supply, electricity, transportation).	At-site or at-source valuation of intermediate goods off-stream (agriculture, industry) and instream (hydropower, transportation, waste assimilation).
Damage costs	No	Indirect use	Maximum WTP given as monetary value of damages avoided.	Valuation of reduced water pollution or flood damages.
Residual value/Farm crop budget	No	Direct use	Constructed models for deriving point estimate of net producers' income or rents attributable to water via budget or spreadsheet analysis.	At-site or at-source estimates for off-stream intermediate goods (agriculture, industry) for single product case.
Mathematical programming	Yes	Direct use	Constructed residual models for deriving net producers' rents or marginal costs attributable to water via (usually) fixed price optimisation models.	At-site or at-source valuation of off-stream intermediate goods (agriculture, industry) for multiple product, multiple technology cases.
Observation of water market transactions	Yes	Direct & indirect use	Observed prices from transactions for short-term leases or permanent sale of rights to water.	Actual at-source or at-site WTP manifested by transactions within/between agricultural, industrial, municipal, and environmental uses.
Production function approach (RP)	Yes	Direct use	Primary or secondary data on industrial and agricultural inputs and outputs analysed with statistical techniques.	Producers' (agricultural or industrial) at-site valuations.
Hedonic property value (RP)	Yes	Direct & indirect use	Uses econometric analysis of data on real property transactions with varying availability of water supply or quality.	At-source demands for changes in water quantity or quality revealed in sales transactions in residential or farm properties.
Travel cost method (RP)	Yes	Indirect use	Uses variations in visitor travel costs and econometric analysis to estimate the demand for recreational site attributes. From the demand curve WTP is calculated.	In water-related recreation from which at-source valuations for changes in water levels or water quality.
Contingent valuation (SP)	Yes	Use & non-use	Uses statistical techniques for analysing responses to survey questions asking for monetary valuation of proposed changes in environmental goods or services.	At-source valuations of recreational or environmental (e.g. instream) of *in situ* water or water quality. Also at-site valuations of changes in residential water supplies. Can measure non-use values.
Discrete choice experiments (SP)	Yes	Use & non-use	Uses statistical techniques to infer WTP for goods or services from survey questions asking a sample of respondents to make choices among alternative proposed policies.	At-source valuations of recreational or environmental (e.g. instream) of *in situ* water or water quality. Also at-site valuations of changes in residential water supplies. Can measure non-use values.

Source: adapted from Young and Loomis, 2014, p.44/5.

2.1.8 Benefit transfer

Defra (2007, p.38) define benefit transfer (also known as value transfer) as 'a process by which the economic values that have been generated in one context – the 'study site' – are applied in another context – the 'policy site' – for which values are required' (this distinction between study and policy sites is an important one that will be relied on throughout what follows). Whilst not an economic valuation technique itself, benefit transfer (BT) is quicker and less expensive than undertaking a primary valuation study using one of the techniques referred to above. Indeed, as indicated in Figure 2.5, in principle, BT values can be derived from all economic valuation techniques irrespective of whether or not they are based on the demand curve.

There are four principal means by which values van be transferred from a study to a policy site as shown below in Table 2.4.

Table 2.4 Benefit transfer methods

Benefit transfer method	Description	Explanation	Example studies
Single point value transfer	A single average WTP estimate is transferred without adjustment from study to policy site.	A wetland protection value of £50 per person is transferred from study site A to site B.	Prokofieva *et al.* (2011), Kubiszewski *et al.* (2013).
Marginal point value transfer	A single value that allows for site differences is transferred.	A wetland protection value of £2 per hectare per person is transferred from study site A to site B. The values are adjusted for the size of the area protected.	Fetene *et al.* (2014).
WTP function transfer	Coefficients, which describe the relationship between WTP and the factors influencing it at the study site, are applied to data from the policy site.	A wetland valuation function that involves several attributes is transferred from case study site A to site B.	Loomis (1992), Fetene *et al.* (2014).
Meta-value analysis	Results of several studies are combined to generate a pooled model.	Results from studies A,B,C and D are pooled to estimate value for site E.	Oglethorpe *et al.* (2000), Rosenberger and Loomis (2000), Shrestha and Loomis (2003), Bergstrom and Taylor (2006), Brander *et al.* (2012).

Source: adapted from Morrison and MacDonald, 2010, p. 19.

The general consensus in the literature is that function transfers (stand-alone or meta-analytic), outperform other methods (e.g. Rosenberger and Stanley, 2006), particularly when the transfer involves dissimilar sites but similar goods (Bateman *et al.* 2009). However, when transferring across similar goods and sites – where factors such as the scope of the change, availability of substitutes and income constraints closely correspond – then single point value transfer may be sufficient (*Ibid*). When transferring values internationally a number of special considerations apply (Ready *et al.* 2004; Czajkowski and Scasny, 2010; Johnston and Rosenberger, 2010). These include considerations of the appropriate exchange rates to use and the necessary adjustments to reflect any disparities in income between the country where values are sourced from and the country in which they are applied.

2.1.9 Existing water valuation studies

Table 2.5 below shows existing empirical valuation studies, on the general topic of water, that are held in the *Environmental Valuation Reference Inventory* (2011) (EVRI) database.[3] These studies encompass values for in-stream and off-stream water uses, and the values listed, crucially here, are in varying denominations. The aim of EVRI is to collate valuation research, compiled on the basis of economic valuation methods, for a variety of environmental goods and services, in order to aid policy analysts in the application of benefits transfer. Whilst EVRI is not exhaustive – there are additional overlapping databases such as *ValueBase SWE*, *The New Zealand Non-Market Valuation Database*, *TEEB* and *Envalue* – it offers the broadest coverage of any similar tool available and thus gives an indication of the number, type and coverage of water related valuation studies that have been conducted to date (see McComb *et al.*, 2006 for a review of the relative coverage of the databases).

The key point here is that, of the 1,735 water valuation studies in EVRI in 2015/6, and the limited number of additional studies not captured by the database, as far as the author is aware, none of these have attempted to estimate the full value (or TEV) of virtual water across supply chains. Indeed, empirical valuation work, based on genuine welfare economics underpinnings, has to date predominantly been on a local scale (i.e. at the

[3] EVRI is a joint initiative which was set up by DEFRA, the US Environmental Protection Agency, Environment Canada, and the Department for Sustainability, Environment, Water, Population and Communities of the Australian Government.

level of the water catchment) and it frequently focuses on the assessment of one aspect of TEV (see for example Oglethorpe and Miliadou, 2000).[4]

Table 2.5 EVRI environmental valuation water research

Water General	Actual Market Pricing	Revealed Preference	Stated Preference	Total	% of overall total
Africa	25	8	23	56	3.2%
Asia	55	24	92	171	9.9%
Caribbean	8	3	18	29	1.7%
Central America	7	2	14	23	1.3%
Europe	56	70	269	395	22.8%
North America	157	289	402	848	48.9%
Oceania	25	28	134	187	10.8%
South America	9	3	14	26	1.5%
Total	342	427	966	1,735	100%
% of Total	19.7%	24.6%	55.7%	100%	

Source: EVRI, 2011.

There are however three recent additions to the grey literature which are of direct relevance in this context. First, Trucost, a UK based environmental data consultancy, have undertaken work for Novo Nordisk and Puma that has fed into the development of their environmental profit and loss accounts, where they have placed monetary values on water use, greenhouse gas emissions and air pollution across the supply chain (PUMA, 2010; Danish Environmental Protection Agency, 2014a; Danish Environmental Protection Agency, 2014b). More recently, the approach adopted to the valuation of water in the supply chain has fed into the Water Risk Monetizer tool that Trucost have developed in conjunction with Ecolab (2015). However, this work suffers from several limitations. First, the valuation approach adopted – which would be classed as a meta-value analysis according to Table 2.4 – focuses solely on in-stream ESS and completely neglects the value associated with off-stream water use. Second, it neglects the value of green and grey water, both of which are terms that will be covered in more detail in the next section of this literature review, but which refer to the value of precipitation and water pollution respectively. Third, the meta value approach adopted by Trucost appears to have been founded predominantly upon the unit values in one article – Frederick *et al.* (1996) – which is itself a meta-analysis of the unit values of water that had been estimated in the USA up until the mid-1990s. Fourth, in assigning a value to in-stream water uses, Trucost utilise water scarcity as the single predictor

[4] Water catchment, river basin and water shed are used synonymously in this thesis.

variable in their regression modelling. However, whilst intuitively appealing and not devoid of theoretical basis, it is nonetheless not grounded in a strict theoretical framework. Finally, the approach by Trucost appears to be aggregated at the business level and is not specific to a certain product. Therefore, it does not focus on a particular supply chain and look to understand how the variations in the value of virtual water might impact on decisions regarding economic efficiency.

In addition to the work by Trucost, a second relevant contribution in the grey literature has arisen from a partnership between the Natural Capital Declaration, German Association for Environmental Management and Sustainability in Financial Institutions and Deutsche Gesellschaft für Internationale Zusammenarbeitand. Specifically, these organisations have produced a model which looks to estimate the TEV of water in different locations for the purposes of corporate bond credit analysis (Ridley and Boland, 2015). However, this model has been applied by Bloomberg to analyse water risk in mining equities (Park *et al.* 2015) and it is claimed that the model has much broader application, potentially including the use of water at multiple sites of interest as would be the case in a supply chain. The basis of the valuation approach adopted is that water not consumed by a company at one of its sites could instead be used for agricultural purposes, municipal supply, the promotion of human health, and by the natural environment. Indeed, each of these water uses is treated as a dependent variable in a meta-analysis, the predictor variables for which are water stress (in all cases) and population within 50 kilometres of each company site (for all water uses except agriculture). However, again, there are several limitations with this approach. First, the values for agriculture water use are based on a meta-analysis of the available literature between 2000 and 2015 only and it is not clear what countries and regions this includes. Second, the values for domestic supply are based on a water price data set which has been sourced from the Global Water Intelligence 2016 Water Price Survey, in conjunction with the simple assumption that price rises with scarcity. However, as alluded to already, the concepts of price and value are not synonymous. Indeed, as indicated in sections 2.1.3 – 2.1.4, the former is often a poor substitute for a real measure of WTP. Third, the values attributable to human health and environmental impact are based not on a measure of WTP, but they have been estimated using impact factors developed by Pfister *et al.* (2009) for use in Life Cycle Analysis. As such they do not represent true welfare measures, a fact which the authors appear to acknowledge in their description of the model as a 'hybrid'. Finally, the most obvious limitation with the model is that it is looking to estimate the value of water in industry with reference to other water uses rather than attempting to estimate what the direct use value of the water is itself in industry. In other words, the value of water is effectively an opportunity cost i.e. the value that would have accrued had it not been consumed in industry. Whilst this may or may not be an acceptable approach, it is making several assumptions about the uses of water in an area (i.e. that it is used in agriculture, municipal supply etc.) which may not be confirmable at the level of spatiotemporal detail that the model appears to operate at.

The third and final contribution in the grey literature that is relevant here has been provide by Veolia and their True Cost of Water Model. This looks to estimate the direct costs, indirect costs and costs related to risks associated with a company's water supply dependence. However, as the names imply, this model is focused on cost rather than value. Indeed, as we have seen, whilst costs can be used as a lower bound estimate of value in non-demand curve based valuation techniques, the aim of the Veolia model, whilst important, is not the estimation of a measure of the true value of water within a TEV framework.

More generally, however, the three models mentioned in the grey literature, to varying degrees, all suffer from the fact that they do not, as far as the author is aware, lay bare all the assumptions that they are based on. Indeed, the nature of the monetary values that they utilise and any exclusion criteria that have been applied to these, the means by which the values have been updated, converted to a common currency (if necessary) and standardised, are all not fully clear. What is more, explanatory variables are deployed in meta-analytic BT exercises without a firm theoretical foundation, and the outcomes of the regression models are not fully described. Therefore, the estimation of the TEV of virtual water within a supply chain, and the extent to which the existing body of environmental valuation literature will support its estimation, together with the assumptions that need to be made in order to operationalise this, remain a fertile research question of note here. However, whilst the models described above are not beyond criticism, the fact that industry has recognised the merit of applying shadow values to water use by companies, and in a supply chain context in the case of Trucost, clearly highlights the importance of this as a research subject to be explored within academia.

Separate to these three important sources in the grey literature, two further bodies of work should also be briefly mentioned here. The first of these is input/output (I-O) modelling which is finding increasing favour within business and management circles and which, at first glance, may suggest that it is doing something similar to the valuation of water use by business, sometimes in a supply chain context (see for example Acquaye *et al*. 2017). However, Young and Loomis (2014, p.86) make the point that I-O models are based on the concept of *value added* which is not an appropriate measure of WTP. Indeed, the value-added approach, 'rather than isolating only the contribution of one input (water), ...imputes the productivity of *all* primary resources (labour, management, entrepreneurship, capital, land and other natural resources, taxes, and even depreciation) to the residual (value of water)' (*Ibid*). As a result, the figure generated by the value-added approach greatly overstates WTP and is thus not relevant here.

The second body of work worth mentioning is natural capital accounting. This has grown in significance in the UK following the work of the Natural Capital Committee and there have been efforts to integrate the value of environmental stocks and flows into national accounts (Obst *et al.*, 2016) and corporate accounts (Eftec *et al.*, 2015). In a water context, the most prominent manifestation has been the System of Environmental Economic Accounting for Water or SEEA which has been proposed by the UN Department for Economic and Social Affairs (UN DESA, 2012). However, to date the natural capital accounting literature has been primarily focused on methods and procedures (e.g. which values are appropriate for inclusion in national accounts and how can they be aggregated at the national level) rather than application. In addition, though, whilst there is a degree of overlap with the focus of this thesis, the natural capital literature ultimately is not, as here, looking to utilise economic values of water resources to inform their efficient utilisation. Moreover, the goal of the literature is to aggregate values at a national or company level and not look at the value of water in a product supply chain context.

2.2 Economic policy instruments – internalising externalities

Section 2.1 noted that the presence of externalities was one of the principal market failures associated with water resources, the effect of which, can be economically sub-optimal allocation. This section now moves on to address how negative externalities can be internalised, or, put another away, how the divergence between private motives and social objectives depicted in Figure 2.2, can be eliminated. What follows is not meant to provide exhaustive coverage of every incentive design that looks to achieve this end, owing to the breadth of the literature involved (for an overview see Hanley *et al.*, 2007). Rather, it aims to provide an overview of the central *economic* policy instruments, and the pervading theoretical currents within, and how they have been applied in a water resources context. This is appropriate here because, as argued previously, the full value of virtual water has not been addressed comprehensively in the academic literature to date, and neither, by extension, has how economic policy instruments might be designed in response to this.

Perhaps the most well-known incentive design in this context is the idea of Pigouvian taxation which was first put forward by Arthur C. Pigou in *The Economics of Welfare* (1920). The central contention here is that an externalities tax, equal to the divergence between MPC and MSC in Figure 2.2, will ensure that market price reflects the true social costs of production, and, that firms are subject to the full social costs of their activities. A slight variation on the Pigouvian theme, pollution permits and tradeable water rights focus on regulating the quantity of the externality as opposed to its price. Both of these approaches can be seen in Figure 2.6 below. In the right-hand diagram, the permit system imposes a fixed supply of the externality, which in conjunction with the demand curve, determines the price. In the left-hand diagram, the Pigouvian tax and the ensuing fixed price, in conjunction with the demand curve, determines the quantity of the externality

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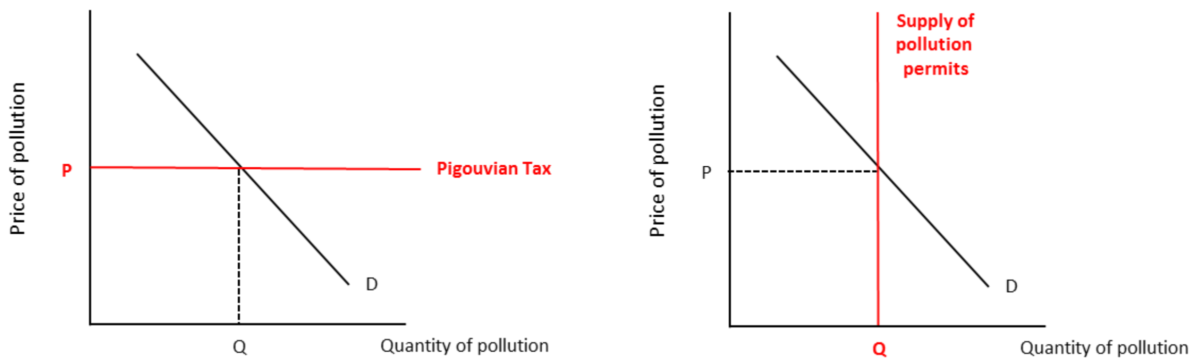

Figure 2.6 Pigouvian taxes (left) and marketable pollution permits/water rights (right).

Both pollution permits and water rights, which entitle the user to pollute or deplete a certain amount respectively, can be and are traded on markets, the effect of which, is to harmonise and minimise total industry wide externality abatement costs.[5] Pollution permits have been used, for example, by the USA's Environmental Protection Agency to improve water quality, and rights for irrigation water are currently traded, on a large scale, across Australia.

Both environmental taxes and permits/rights are based on the Polluter Pays Principle (PPP) which was formally adopted by the OECD in 1972 and has also been incorporated into EU treaties (Ekins 1999). The aim of PPP is to ensure that those responsible for any externalities bear the associated abatement costs. However, PPP has been challenged, particularly in the UK and USA, by what might be called the Provider Gets Principle (PGP) (Pretty *et al.*, 2001). Rather than privileging the fundamental property rights of the state as in PPP, PGP emphasises the vested property rights of land owners and thus advocates the use of public subsidy in order to achieve environmental outcomes (*Ibid*). In other words, rather than seeking damages from land owners, PGP favours offering them compensation if environmental goals negatively impact their profits (Hanley and Oglethorpe 1999).

Finally, in direct challenge to the work of Pigou, Coase (1960) proposed a market based approach in contrast to the interventionist instruments mentioned above. The Coase Theorem, as this came to be known, advocated that if property rights could be made explicit and freely transferrable, and if transaction costs were limited, private parties would be able to bargain over the allocation of resources, and in so doing, solve the issue of any associated externalities on their own. Indeed, this process of market mediated bargaining, it was argued, would deliver the optimum level of the externality by ensuring that property rights accrued to their highest valued use, irrespective of any initial allocation. The only role for government then in this scenario is to assign and enforce these property rights.

The Coase theorem has been applied to water pollution by Dales (1968) and it was later extended and developed by Baumol and Oates (1971) (*quoted in* Tietenberg, 2010).

[5] In the example of pollution permits, polluters with high marginal abatement costs relative to the price of the permits, have an incentive to buy. Conversely, those polluters with low marginal abatement costs relative to the price of permits are incentivised to sell. The process of buying and selling, and the resulting equilibrium price, equates marginal abatement costs across firms.

2.3 Virtual water

This section begins by briefly setting out the origins of the virtual water concept, before introducing the idea of the water footprint and water footprint assessment that emerged from this. Subsequently, the evolution, application and criticisms associated with water footprint research, are presented.

2.3.1 The origins of the concept

As originally conceived, the idea of associating products with their inputs, inherent in virtual water, was titled *embedded water* when first used by Allan (1993; 1994 *quoted in* Allan, 2003). This idea had been inspired by Israeli economists of the mid-1980s who had 'spotted that it was less than sensible from an economic perspective to export scarce Israeli water' in the form of water intensive oranges or avocados (Allan, 2003, p.4). Later re-titled *virtual water* (Allan, 1996; 1998; 1999), the distinct focus on the international trade in agricultural crops and how this enabled water disadvantaged regions – and in particular the MENA region – to attain food security by importing water intensive produce from comparatively advantaged regions, remained. Indeed, in spite of the fact that embedded/virtual water did not emanate from the economics literature, as Reimer (2012, p.135) suggests, it is an inherently economic concept, and one which has close – albeit contested – links to the idea of comparative advantage. Indeed, much of the ongoing debate about virtual water surrounds whether, conceived as a factor of production, it is a determinant of international trade and thus susceptible to analysis by the Heckscher-Ohlin model, or whether it is simply an engaging metaphor (e.g. Allan, 2003; Merett, 2003; Wichelns, 2004; Ansink, 2010; Hoekstra, 2010; Reimer, 2012).

2.3.2 The water footprint

The principal legacy of the virtual water debate, however, has been that it directly fed into the *water footprint* concept which was first introduced by Hoekstra (2003), to the extent that the two terms have largely now become synonymous, and indeed will be used as such in this thesis. Introducing supply chain thinking into the water studies discipline, the water footprint extends and develops the notion of virtual water by providing a full methodology for the analysis of virtual water flows known as water footprint assessment (WFA) (in the following chapters WFA will also be referred to generically as 'water footprinting'). Indeed, whilst the water footprint accounts for both the volumes of direct (i.e. operational) and indirect (i.e. supply chain) water use, it also specifies the type of water used and its geographical and temporal distribution, thereby going beyond the more simplistic, volume focus, of embedded water.

Water footprinting recognises three types of water use, each of which comprises the direct and indirect water footprint:

1. Blue water – the consumption of surface and ground water.
2. Green water – the consumption of rainwater, stored in the soil as moisture (i.e. not lost in run–off or ground water recharge), during the production process.
3. Grey water – 'the volume of polluted water defined as the volume of freshwater that is required to assimilate the load of pollutants given natural background concentrations and existing ambient water quality standards' (Hoekstra *et al.*, 2011, p.2).

Reference to consumptive water use is important here. In the context of blue water, this refers to losses of ground or surface water from a catchment area when 'water evaporates [in the course of production], returns to another catchment area or the sea, or is incorporated into a product' (Hoekstra *et al.*, 2011, p.2). In a green water context, consumption refers to total rainwater evaporated or incorporated into a product. In conjunction, the green and blue footprints make up the consumptive water footprint. Consumption does not mean that water vanishes; water is a renewable resource. However, its availability during a certain period is limited and the consumptive water footprint indicates the volume of water not immediately available for other uses. Moreover, it is 'particularly consumptive use that determines the impact on the water system of a catchment' (Hoekstra *et al.*, 2011, p. 74).

When compared to traditional measures of water use which look simply at 'water withdrawals,' the water footprint differs in three key respects (Hoekstra *et al.*, 2011, p.3):

1. It does not include blue water use insofar as this water is returned to where it came from and is thus available for other uses.
2. It is not restricted to blue water use, but also accounts for green and grey water.
3. It is not restricted to direct water use, but also includes indirect water use.

2.3.3 Water footprint assessment

The water footprint concept – i.e. the spatially and temporally explicit analysis of the direct and indirect use of blue, green and grey water – can be applied to a single process, product, business, consumer, group of consumers (e.g. consumers in a nation), or a specific geographical area such as a country. However, the basic building block of WFA is the single process step. In the context of a product water footprint, which is the subject of interest in this thesis, this is made up of the relevant process steps that occur in direct production of the product (also known as the *operational footprint*). However, the operational footprint constitutes only one of four elements that together make up the product water footprint. In conjunction with the *supply chain water footprint* (2), which refers to the water footprint associated with the ingredients and other inputs that go into making the product, both of these components are said to be *directly associated* with the inputs that are used to produce the product. Indeed, this reference is an important distinction that will be relied upon in subsequent chapters. By contrast, the s*upply chain overhead water footprint* (3) 'originates from the all goods and services used in the factory that are not directly used in or for the production process of one particular product produced in the factory' (Ercin *et al.,* 2011, p.727). Similarly, the *operational overhead water footprint* (4) 'refers to freshwater use that … cannot be fully associated with the production of the specific product considered, but refers to freshwater use that associates with supporting activities and materials used in the business, which produces not just this specific product but other products as well' (Hoekstra *et al.,* 2011). Therefore, it is by estimating the water footprint associated with the process steps used to produce the product, together with the supply chain, operational overhead and supply chain overhead water footprints, that the product water footprint is derived.

The process of *water footprint assessment* however goes beyond simply accounting for water volumes which has been the subject of discussion so far, and consists of four phases: 1) setting scope and goals, 2) water footprint accounting, 3) water footprint sustainability assessment, and 4) water footprint response formulation (Hoekstra *et al.,* 2011, p.4). As can be seen, beyond phase two, assessment also endeavours to gauge the potential social, environmental and economic impacts of the water volumes calculated, and their sustainability (phase three), as well as design appropriate policy responses (phase four).

In phase three (assessment of sustainability), WFA poses two key questions for each component of a water footprint (Hoekstra *et al.,* 2011, p. 92)[6]:

1. Geographic context – is the water footprint component located in a catchment area and period of the year that was identified as a hotspot?
2. Characteristics of the component – is the water footprint of the process itself unsustainable: in other words, can the water footprint be avoided altogether or reduced at reasonable societal cost?

On the first of these, hotspots are identified in economic, social and environmental terms, each with their own sustainability criteria as shown in Table 2.6 below.

Table 2.6 Sustainability criteria for identifying hotspots

Hotspot type	Sustainability criteria
Environmental	Are there periods of time within a catchment when 'environmental green or blue water needs or water quality standards are violated?' (Hoekstra *et al.,* 2011, p. 87). A green water footprint forms an environmental hotspot if it exceeds the availability of green water. Similarly, if the blue water footprint exceeds blue water availability and/or 'results in a drop in groundwater or lake levels to an extent that these drops exceed a certain environmental threshold,' this represents a blue water hotspot (Hoekstra *et al.,* 2011, p. 79) Finally, a grey water footprint forms a hotspot when 'ambient water quality standards in that period in that catchment are violated, in other words, when waste assimilation capacity is fully consumed' (Hoekstra *et al.,* 2011, p. 86).
Social	Are basic human needs and basic rules of fairness being met? Assessment of the former is based on access to a 'minimum amount of safe and clean freshwater supply for drinking, washing and cooking and a minimum allocation of water to food production to secure a sufficient level of food supply to all' (Hoekstra *et al.,* 2011, p.77). The latter is determined by the proper compensation of downstream users by upstream water users and polluters in the form of the water user pays and the polluter pays principles. An additional fairness criterion is the fair consumption of public goods (Ibid, p.87).
Economic	Is water being i) allocated, and ii) used efficiently? Assessment is based on the degree to which full costs (defined as externalities, opportunity costs and a scarcity rent) are charged to water users (Hoekstra *et al.,* 2011, p. 88).

The *Water Footprint Assessment Manual* provides 'an inventory of options' for phase 4 and the formulation of consumer, producer, investor and governmental responses to the sustainability assessment phase (Hoekstra *et al.,* 2011). However, it is purposely not prescriptive and as a result it does not articulate what to do or how to do it. This being said, it is noteworthy in this context that one of the governmental response options listed is to 'restructure water pricing mechanisms such that full costs of water inputs become

[6] A component refers to 'one specific process [which] occurs in a specific part of the year in a specific catchment' (Hoekstra *et al.,* 2011, p. 91).

part of the cost of final commodities,' even if there is not a suggested means to achieve this (Hoekstra *et al.*, 2011, p. 113).

2.3.4 The evolution of water footprint research

Comprehensive guides to the evolution of water footprint research are provided by Zhang and Hoekstra (2013), and Zhang *et al.* (2017) who undertake a full bibliometric analysis of the literature. However, the main aspects of this evolution, which is best characterised as occurring in three principal stages, are presented below.

In stage one, early water footprint research focused predominantly on accounting for the volumes of green and blue water in processes, products and companies rather than sustainability assessment and response formulation. Two seminal papers in this regard, Hoekstra and Hung (2002) and Chapagain and Hoekstra (2004), between them, developed global statistics encompassing the water footprints of a wide range of crops, animal products, domestic and industrial sectors, and the flows of trade induced virtual water. To these first generation *meta* papers, titled as such here due to the extent of their coverage, Chapagain *et al.*,(2006) developed the notion of dilution volume which later became the grey water footprint. At around this point, in October 2008, the Water Footprint Network was formally established in Enschede, The Netherlands, with the aim of promoting WFA and through it, the sustainable use of freshwater.[7]

In stage two, drawing on methodological advancements that were not incorporated in the first generation *meta* papers referred to above, Mekonnen and Hoekstra were able to more explicitly distinguish between different forms of water consumption and provide greater spatial and temporal definition, in estimating water footprints for agriculture, farm animal products and industry (Mekonnen and Hoekstra, 2010a; Mekonnen and Hoekstra, 2010b; Mekonnen and Hoekstra, 2011; Mekonnen and Hoekstra, 2012a). Adding to the coverage of these second generation *meta* papers, Dominguez-Faus *et al.,* (2009), Gerbens-Leenes *et al.*, (2009) and Mekonnen and Hoekstra (2012b), have analysed the water footprint of the renewal energy sector to include biofuels, biomass and hydropower.

[7] This is a joint endeavour between the University of Twente, WWF, UNESCO-IHE, World Business Council for Sustainable Development, International Finance Corporation, Netherlands Water Partnership, and Water Neutral Foundation.

These advancements were formalised in the development of the *Global Water Footprint Standard* – as set out in the *Water Footprint Assessment Manual* – in order to ensure methodological rigour, and with it, accuracy of comparison between different WFA studies (Hoekstra *et al.,* 2011). The creation of an on-line *Water Stat* database in order to provide greater access to water footprint data, and the development on an on-line *Water Footprint Assessment Tool*, further solidified this common approach (Water Footprint Network n.d. a; Water Footprint Network n.d. b). Both the *Water Stat* database and the *Water Footprint Assessment Tool* are predominantly populated by data from the second generation *meta* papers referred to above. However, they have also been informed by a number of product and geographically specific studies as shown in Tables 2.7 and 2.8 below.

Table 2.7 Discrete product water footprint studies

Product	Publication
Bread	Mekonnen and Hoekstra (2010c).
Coffee and tea	Chapagain and Hoekstra (2007).
Cotton	Chapagain *et al.* (2006).
Pasta and pizza	Aldaya and Hoekstra (2010).
Rice	Chapagain and Hoekstra (2011).
Soft drinks	Ercin *et al.* (2011).

Table 2.8 Discrete geographic water footprint studies

Country	Publication
China	Ma *et al.* (2006); Liu *et al.* (2007); Liu and Savenije (2008); Hubacek *et al.* (2009); Zhao *et al.* (2009); Ma *et al.* (2015)*
France	Ercin *et al.* (2013)
Germany	Kumar and Jain (2007); Sonnenberg *et al.* (2009).
India	Verma *et al.* (2009).
Indonesia	Bulsink *et al.* (2009).
Morocco	Hoekstra and Chapagain (2007).
Netherlands	Hoekstra and Chapagain (2007); van Oel *et al.* (2009).
Romania	Ene and Teodosiu (2009).
Spain	Aldaya *et al.* (2008); Novo *et al.* (2009).
UK	Chapagain and Orr (2008).
USA	Rushforth and Ruddell (2015)**

* Water footprint study of Beijing. ** Water footprint study of Pheonix, Arizona.

More recently, in stage three, water footprint research has begun to go beyond simple water accounting. For example, Francke and Castro (2013) and Hoekstra and Wiedmann (2014) have focused on how different footprinting concepts (land, water, energy etc.) can be applied in conjunction. Seekell (2011), Hoekstra (2014a), Mekonnen and Hoekstra (2014) and Chukalla *et al.* (2015), between them, have proposed and set out water footprint caps per river basin, water footprint shares per community, and water footprint benchmarks per product, as a means of sustainably and equitably allocating freshwater resources. Gerbens-Leenes (2013) and Hoekstra (2014b) have examined the significance of the livestock sector, and in particular livestock feed, to humanity's water footprint and how changing consumption patterns effect this. Ercin and Hoekstra (2014) have set out four different water footprint scenarios that might prevail in 2050 depending on certain key demographic and socio-economic drivers. This followed assessments which have quantified blue water scarcity in over 400 river basins (Hoekstra and Mekonnen, 2011), and estimated past and future trends in grey water footprints of anthropogenic nitrogen and phosphorous inputs into the world's main river catchments (Liu *et al.*, 2012). Grey water footprints related to historic nitrogen loads have been further elaborated, at higher levels of spatiotemporal detail, by river basin, economic sector and crop type (Mekonnen and Hoekstra, 2015).

2.3.5 Application of the water footprint – business and policy

In a business context, a wide variety of companies across multiple sectors have implemented WFA as part of their CSR profile and as a means of tackling water related business risks. Table 2.9 below shows the results which have been published to date.

Table 2.9 Industry application of WFA

Company	Industry	Publication
Barilla[8]	Food and beverage (Pasta)	Ruini *et al.* (2013a); Antonelli and Ruini (2015)
Beverage Industry Environmental Roundtable (BIER)[9]	Food and beverage	Beveridge Industry Environmental Round Table (2011).
Coca-Cola	Food and beverage (Coca cola and orange juice)	The Coca Cola Company and The Nature Conservancy (2010); Coca Cola Europe (2011).
Dole	Food and beverage (Bananas and pineapples)	Sikirica (2011).
Mars	Food and beverage (Sweets and pasta sauce)	Ridoutt *et al.* (2009)
Natura[10]	Cosmetics (Soap)	Francke and Castro (2013).
Nestle	Food and beverage (Breakfast cereal)	Chapagain and Orr (2010).
SAB Miller	Food and beverage (Beer)	SAB Miller and WWF-UK (2009); SAB Miller, WWF-UK, and GTZ (2010).
Tata	Manufacturing	IFC, TATA Group, and Water Footorint Network (2013).
Unilever	Food and beverage (Tea and margarine)	Jefferies *et al.* (2012).
UPM-Kymmene	Paper	Rep (2011).

[8] WFA used in the broader context of an LCA study.
[9] BIER consists of, amongst others, Ocean Spray, Pepsico, Nestle Waters, Danone Waters, Barcardi, Carlsberg Group, Diageo, Heineken, The Coca-Cola Company, Miller Coors.
[10] WFA used in the broader context of an LCA study.

Echoing the evolution in WFA research mentioned above, the use of WFA by industry has evolved from a situation where early studies were emerging predominantly from the food and beverage sector (with the agricultural focus of their supply chains) and focusing on phase two and accounting for water volumes (e.g. The Coca Cola Company and The Nature Conservancy, 2010). More recently, WFA has, to an extent, expanded to include phase three and four and spread to sectors with little or no association with agriculture, the most telling being the case of Tata Steel (see IFC *et al.,* 2013). However, at present, the bulk of practical applications for WFA in industry remain focused on accounting for volumes, to the extent that WFA recognises that when 'more practical applications become available, this will provide valuable inputs for refining procedures and methods [for sustainability assessment]' (Hoekstra *et al.,* 2011, p.118). Crucially in this context though, whilst the volume accounting pursued by the companies in Table 2.9 does include the full supply chain geographical distribution of blue, green and grey water, no work has been published to date on how to extend this to include a notion of full economic and societal value as proposed here.

Some of the key themes that have emerged from the interaction between industry and WFA have included the following:

1. Confounding traditional approaches which look to assess water usage in direct operations, indirect water usage is often many times larger, particularly when there is an agricultural aspect to a supply chain. Indeed, in the case of Coca-Cola, 99% of its water footprint is associated with supply chain ingredients (The Coca Cola Company and The Nature Conservancy 2010; Ercin *et al.,* 2011).
2. The majority of water withdrawn by industry gets returned to the same basin i.e. it does not count towards the consumptive water footprint. As Hoekstra and Mekonnen (2012) show, only 3.7% of the global blue water footprint is attributable to industry.
3. In spite of point two above, industry accounts for 26.3% of the global grey water footprint, suggesting that while it does not consume as much as it withdraws, the water that is returned is not adequately treated (Hoekstra and Mekonnen, 2012).

By comparison to the impact of WFA in the business world, its impact in formulating policies and water management decisions, has been far more limited. The work of Aldaya and Llamas (2008), and in particular, their focus on the economic assessment of water footprints in the Guadiana basin, has informed the EU Water Framework Directive (WFD) assessments in Spain. Indeed, in 2010, the Spanish government introduced a regulation that requires WFA to be used as a tool for the implementation of River Basin Management plans prescribed by the WFD (Aldaya *et al.,* 2010b). However, these examples aside, WFA has not found wide and explicit policy application.

2.3.6 Criticisms of WFA

To begin with here, a number of criticisms can be levelled at WFA on its own terms. First, as indicated, sustainability assessment is a relatively new area of WFA and little work has been conducted to date on the development, and certainly implementation, of what is limited methodological guidance on the identification of hotspots. Indeed, part of this likely stems from the fact that whilst phase two and accounting for water volumes can be conducted at low levels of spatiotemporal detail and thus can be used for an initial analysis, many of the tools for sustainability assessment which seem to be the focus of WFA scholars (water footprint benchmarks, water basin caps and ad hoc deliberations regarding water footprint shares per community) require far higher levels of prior knowledge and thus are less accessible. Second, the second question that WFA posits in phase three (i.e. characteristics of each component) refers to a 'reasonable' societal cost of reducing or eliminating a particular process. However, again, WFA is not prescriptive as to how to assess 'reasonable' costs or the ensuing benefits against which trade-offs such as these are judged.[11] Furthermore, economic hotspots are defined with reference to the extent to which full costs are charged to water users, however, again the measurement and assessment of these costs is not prescribed. All of these are issues which, in principle, could be addressed by supplementing WFA with notions of full economic and societal value, provided that sufficient values are available, which is why this remains a potential research question of note here.

Criticisms by omission aside, the main critique of WFA emanating from the academic literature have come from the Life Cycle Analysis (LCA) discipline which is accustomed to carbon footprints which can be expressed as a single figure (carbon dioxide equivalents or Co_2-e) and thus easily compared (Pfister *et al.,* 2009; Ridoutt *et al.,* 2009;

[11] The Water Footprint Assessment Manual recognises that in 'internalising economic and environmental externalities posed by [the] overexploitation and pollution of water, water footprint reduction will generally result in a societal benefit, or at most, a reasonable societal cost' (Hoekstra *et al.,* 2011, p. 90). However, as mentioned, WFA does not prescribe a practical means to achieve this.

Ridoutt and Pfister, 2010). As such, the purely 'volumetric' nature of WFA has been characterised as a 'crude summation of more than one form of water consumption from locations that differ in terms of water scarcity' (Ridoutt and Pfister, 2010, p. 114). As a result, water footprints of different products are not comparable, it is argued, and they do not denote potential social or environmental harm. As Ridoutt and Pfister (2010, p. 114) suggest, 'it is not clear what good would result from choosing a product or production system on the basis of it having a lower water footprint,' given that 'a product with a lower footprint could be more damaging to the environment than one with a higher water footprint depending on where the water is sourced.' As a means of addressing this, Ridoutt and Pfister (2010) advocate weighting the water footprint by the *Water Stress Index* (WSI) to arrive at a stress weighted water footprint.[12] More recently, Boulay *et al.* (2015) have proposed the *Available Water Remaining* (AWaRe) per m^2 indicator, which refers to the available water in a basin minus the human and environmental water demands, as an alternative water stress indicator for use in weighting water volumes.

WFA has responded to this criticism by including stress weighted water footprints in the latest incarnation of the *Water Footprint Assessment Manual*, for use in looking at the local environmental impacts of products during sustainability assessment (phase three) (Hoekstra *et al.*, 2011). However, in their reply to the LCA community, Hoekstra *et al.* (2009, p.114) argue that whilst this stress weighting makes sense within the logic of LCA with its focus on aggregated impacts and 'characterisation factors,' in a water resources management context, it is necessary to have 'spatially and temporally explicit information on water footprints in real volumes and impacts in real terms.' When this crucial information is removed in the process of creating aggregated impact indices, that which is left is not useful i) as a basis for formulating specific response measures (WFA phase four) and, ii) for discussions of sustainable and equitable freshwater appropriation and allocation (Hoekstra *et al.*, 2011, p. 96). On the subject of the latter, WFA suggests that by ignoring water consumption and pollution in volumetric terms and focusing solely on local environmental impacts, LCA is overlooking the larger issue of global water scarcity. This follows, according to this position, because total appropriation of the globes limited water resources for products is still of paramount importance even though local impacts may potentially differ depending on where this appropriation

[12] The WSI is the ratio of freshwater withdrawals to freshwater availability in different areas.

occurs (Hoekstra *et al.*, 2011, p. 94). From this perspective, the focus shifts to water rich areas producing water intensive commodities and increasing their water productivity because broad trade-offs such as these have the potential to diminish the need to use water for producing those commodities in water scarce areas (Hoekstra *et al.,* 2011, p. 74).

This debate between WFA and LCA is ongoing (Hoekstra, 2016; Pfister *et al.,* 2017) and there have been numerous attempts to integrate the two methodologies which are beyond the scope of this thesis (Milà I Canals *et al.,* 2009; Berger and Finkbeiner, 2013; Chenoweth *et al.,* 2014). However, what it clearly indicates, at least at the local level, is that volumetric analysis in isolation can be insufficient when it comes to assessing impact and comparing water footprints. Therefore, given the link between value and scarcity, valuation has the potential to provide an additional means of assessing water impact. However, by monetising the notion of water impact, it offers a metric which is more understandable and accessible than the myriad of impact categories which are offered by the stress weighted water footprint. At the broader water resources management level, supplementing WFA with the full economic and societal value of water could provide the economic basis, and thus the potential justification, for water rich areas assuming the burden of producing water intensive products as mentioned above. Indeed, it could be argued that it is the very absence of economic values associated with many aspects of water use which provides the rationale for, and interest in, empirical assessments of volumetric virtual water use in the first place i.e. it becomes of interest to map water resources because they are not being allocated efficiently due to the market failures discussed earlier. It is for these reasons that going beyond the volumetric focus of WFA, and the idea of stress weighting water footprints, to include notions of full value remains an interesting and wholly novel potential research question. Indeed, whilst there are nascent indications that the broader academic literature is beginning to recognise the utility of valuation to approaches such as WFA and LCA (for example see Pizzol *et al.*, 2015), as yet this has not progressed to the peer reviewed application of these techniques.

A second criticism of WFA from within the LCA community concerns the boundary of product water footprint assessments. As a number of authors have recognised, it can be informative to include the use and disposal stages of a product's lifecycle rather than just focusing simply on the production stage (Milà I Canals *et al.*, 2009; Ridoutt *et al.,* 2009; Francke and Castro, 2013; Ruini *et al.*, 2013). Whilst these other phases are recognised in WFA, they fall within the footprint of a consumer in the first instance, and a business in the second, and thus represent an additional novel line of research enquiry (Hoekstra *et al.*, 2011, p. 71).

2.3.7 The most robust means of measuring virtual water

Whilst an apparent alternative to WFA has been referenced above (i.e. LCA), it should be noted that the valuation of water resources that is the focus here fundamentally requires an understanding of water in volumetric terms. Valuation cannot be applied to stress weighted volumes. Moreover, historically at least, LCA has lacked an appreciation of water consumption as opposed to withdrawal, and it continues to neglect green water and remains undecided on how best to incorporate grey water (Pfister *et al.*, 2009; Jefferies *et al.*, 2012; Hoekstra, 2016). In addition, water footprint accounting (phase two), *as distinct from WFA*, is the only technique which provides clear and consistent enough terminology to be comprehensible to multiple and non-specialist audiences and thus serve as a pervasive advocacy tool (Chapagain and Tickner, 2012). Indeed, water footprint accounting is an accepted means of calculating the blue water footprint of products in the latest incarnations of the stress weighted water footprint even if subsequent assessment methods differ. It is for these reasons that, in this context, water footprint accounting is the most robust empirical means that is available for measuring virtual water volumes. The contribution of the research that follows will be in the assessment of these water volumes.

2.4 Research questions

In the preceding sections, two primary arguments have been advanced: 1) that the discipline of environmental economics has not embraced the valuation of virtual water, save for three sources in the grey literature which simultaneously highlight the importance of the topic, but also limitations in the methodologies that have been applied, and 2) water footprint research has not embraced environmental valuation despite the fact that it is the absence of a value for virtual water that provides the very rationale for the existence of WFA. Subsidiary to these two arguments, it has also been suggested that monetising virtual water flows would provide an indication of the impact of geographically disparate water usage, but do so in a metric which all stakeholders, and particularly businesses, would understand. Moreover, the valuation of virtual water also the holds the potential to inform intra-supply chain allocative and productive efficiencies.

The principal aim of this thesis then is the marrying of previously unmarried literatures and with it, the development and testing of a method for the valuation of virtual water flows that addresses the gaps identified in previous attempts in the grey literature. In light of this, the four research questions (RQ) that will be pursued in the following chapters are set out below, together with the *principal* sections of the thesis which address them.

- RQ1 Can the existing body of environmental valuation literature support the estimation of unit values of fresh water use that can be transferred to the multiple geographies that global supply chains encompass? [Chapters Three and Seven]
- RQ2 How is the full value of virtual water, within selected supply chain case studies, distributed by: 1) supply chain stage, and 2) geography? [Chapters Four – Seven]
- RQ3 What does the inclusion of a measure of the full value of virtual water reveal about the efficient use and allocation of water in supply chains? [Chapters Four – Seven]
- RQ4 How can regulatory instruments be designed in response to the full value of virtual water and its relative distribution in supply chains? [Chapter Seven]

As shown, RQ1 is looking to establish the feasibility of developing a method, based on the existing literature, that could be used in a benefits transfer exercise that would potentially involve globally disparate regions. This question will be addressed during Chapter Three which sets out the methodology applicable to this thesis. The second and third questions, on the assumption that a method is feasible, aim to address how values vary in a supply chain and what that this tells us regarding efficient allocation within a welfare economics framework. Both RQ2 and RQ3 will be addressed directly in each of the three case studies that are presented in Chapters Four to Six. Finally, RQ4 asks what implications this might have in a policy context, something which will be addressed in Chapter Seven when the conclusions, implications and recommendations from the research project are discussed.

It should be noted here that the use of benefits transfer is presupposed by the aims inherent in RQ1. However, the use of this approach will be fully justified in the following chapter which now introduces the methodology that has been developed and applied in order to address the four RQs identified.

3. Methodology

Parts of this chapter were presented at the 2016 British Academy of Management Conference in a paper titled "A *proposed new method for placing monetary values on virtual water to improve the efficiency of global supply chains*."

The approach to the first RQ posed in this thesis – regarding the scope of the existing body of environmental valuation literature and its potential to enable the estimation of unit water values in spatially disparate regions – is contingent upon both a thorough analysis of the specific values contained in this literature, and a specific framework to guide this analysis. As a result, the methodological approach that has been applied in this thesis has been split into three parts. Part One sets out those broader aspects of the methodology that are not contingent upon the precise method used to value virtual water, starting with a brief discussion of the philosophical position which underpins the research project (section 3.1), before moving on to research design (section 3.2). Following this, the specific methods used to quantify virtual water, together with the broad framework that will be applied in its valuation, will be described (section 3.3). Part Two then builds on the valuation framework set out in section 3.3 by analysing, in detail, the existing body of unit value estimates of water that correspond to the components of the valuation framework. In light of this, Part Three then introduces the specific approach that has been deployed to place values on virtual water in spatially disparate regions.

Part One

3.1. Philosophical position

Given the readily apparent focus of Chapter Two regarding conceptions of what constitutes value, and the means for estimating this, both grounded in welfare economics and the associated axioms of *homo economicus* (represented as rational choice, stable preferences, utility maximisation and market equilibrium), it is quite clear that this research project is underpinned by a positivist philosophical orientation. Whilst alternative paradigms were initially considered here, ultimately, the focus of the opening question posed on page four, and the final RQs arrived at in section 2.4, privilege replicable observations of economic value, in monetary terms, and thus are philosophically in keeping with positivism.

The epistemological and ontological facets of positivism will not be rehearsed here. However, the following sections will nevertheless demonstrate how positivism has infused the methodology and research design adopted.

3.2 Methodology and research design

The research design has three central phases, each of which contributes to an assessment of the full value of the virtual water used in the three case study product supply chains, the results of which, are set out in Chapters Four to Six:

1. *Quantification* of the blue, green and grey water volumes – measured in m^3 – used in direct operations and indirectly in the supply chain, to produce and consume (where applicable) one Stock Keeping Unit (SKU) of the product in question.[13]
2. *Valuation* of the above water use, in monetary terms, to arrive at an estimation of its full economic and societal value and how this is distributed geographically and by supply chain stage.
3. *Reflection* on the implications of points one and two for the RQs and hypotheses laid out, and the bipartite theoretical framework that sits them (i.e. principally, welfare economics and the various incentive designs that stem from this).

The methodological framework described below and in Part Three (centred on the controlled and precise numerical estimation of water volumes and their monetary values, and extensive use of secondary data itself gathered by observation and experimentation) will be generalisable to products and situations other than those in the case studies and thus represents the principal contribution that will stem from the research. However, as point three suggests, in synthesising the results from this methodological advance with the broader theoretical context, the research will also enable us to reflect on the central precepts of welfare economic theory as applied in a wholly novel context i.e. cross border supply chains.

Indeed, the intention here is not to construct theory and nor is it to deduce causal effects which have application beyond the respective case study contexts to a broader population. As such, the broad approach adopted, whilst experimental in nature, might be labelled as a *non-analytic* and *descriptive*, seeking as it does to understand what is

[13] Additional production volumes are also analysed where this informs the analysis.

happening in a particular case at a specific point in time (i.e. the three case studies and the variation in the value of the water within).

3.2.1. Case study selection

Before the specific methods used in the empirical components of the research design are elaborated, a word on the selection of the three case study products – durum wheat pasta, tea and potato crisps – that are examined in this thesis. Two principal criteria – one practical and one methodological – were used to select these products. These were:

I. Supply chain complexity and spatial coverage.
II. The degree to which the product either impacts on, or is impacted by, freshwater resources i.e. the degree to which the product is worthy of study in its own right from a water perspective.

On the first of these, each of the product supply chains chosen necessarily incorporates some secondary manufacture/processing in order to ensure that there are sufficient supply chain stages, across multiple countries, for which values can be estimated and the geographical distribution of these values can be assessed. However, conversely, owing to the difficulty in gaining full supply chain visibility for more complex industrial products which have multi-tiered suppliers, and the time demands associated with undertaking economic valuations for multiple supply chain stages (see Part Three), the supply chains assessed have been selected because they provide the necessary degree of spatial coverage without being overly complex. On the second point, and most importantly, Barton *et al.* (2011) offer a list of those industry sectors that are significantly exposed to water related risks, foremost amongst which, is agriculture. Indeed, owing to the particularly water intensive nature of agriculture – globally 70-80% of freshwater resources are used for this purpose – there are contemporary concerns regarding whether we are 'feeding ourselves thirsty' (Roberts and Barton, 2015). As a result, the three products have been primarily chosen because they are agricultural based supply chains that, whilst underpinned by a variety of crop types (i.e. wheat, potatoes and tea), are all water intensive in nature. The WWF have categorised wheat and potatoes as 'thirsty crops' (WWF, no date) and tea, a predominantly rain-fed crop, is now both increasingly irrigated given climatic change, and grown in areas of increasing water stress (FAO, 2011 and FAO, 2015c).

3.3 Methods

The specific methods that will be adopted to address the two empirical phases of the research design (i.e. the *quantification* and then *valuation* of virtual water) are set out below.

3.3.1 Phase one – Quantification of virtual water

As indicated in Chapter Two, water footprint accounting represents the most comprehensive and robust method when measuring freshwater use in product supply chains and thus will be deployed in this study. The sub-sections below set out precisely how this approach has been implemented, together with several amendments that have been made to water footprint accounting in order to ensure that the volumetric measures are appropriate for valuation purposes in phase two.

3.3.1.1 Sourcing and generating water footprint data - levels of spatiotemporal detail

WFA refers to three distinct levels of spatiotemporal detail which provide a guide for categorising approaches to the collection of water footprint data of all types i.e. not just product water footprints but business footprints, river basin footprints, and the footprints of geographically delineated areas (Table 3.1).

Table 3.1 Spatiotemporal explication in WFA

	Spatial explication	Temporal explication	Source of required data on water use	Typical use of the accounts
Level A	Global average	Annual	Available literature and databases on typical water consumption and pollution by product process.	Awareness raising; rough identification of components contributing most to the overall water footprint; development of global projections of water consumption.
Level B	National, regional or catchment specific	Annual or monthly	As above, but use of nationally, regionally or catchment specific data.	Rough identification of spatial spreading and variability; knowledge base for hotspot identification and water allocation decisions.
Level C	Small catchment or field specific	Monthly or daily	Empirical data or (if not directly measurable) best estimates on water consumption and pollution, specified by location and over the year.	Knowledge base for carrying out a water footprint sustainability assessment; formulation of a strategy to reduce water footprints and associated impacts.

Source: Hoekstra *et al.*, 2011, p. 12.

Given that RQs two and three focus on the variation in monetary values across the *broad* geographies that the three product supply chains span (and associated water allocation decisions), they are not amenable to the level of detail suggested by level C. Level C suggests highly detailed basin and sub-basin specific analysis at each stage along a supply chain. In practical terms, this would prevent the examination of anything but single and geographically bound products. As mentioned in Chapter Two, this explains why the water footprinting literature, for product water footprints at least, has tended to focus on lower levels of spatiotemporal detail at levels A and B. In view of this, an explicit decision has been taken here to found the valuation approach that will be detailed in what follows on a level of spatiotemporal detail which is in accordance, at a minimum, with level B. This will ensure that the valuation methodology developed is generalisable to other products and processes as mentioned earlier, and is thus not limited in application. However, there are two principal implications of this. The first of these, as will be expanded on, is that the methodology developed here is best viewed as a starting point in the analysis of intra-supply chain water allocation decisions, and not a definitive guide. The second is with regard to the water footprint data used in the three case studies. Whilst the case study in Chapter Six on the potato crisp supply chain is based upon empirical data collected directly from the company following discussions with key company personnel and access to internal documentation (thus providing a high degree of spatial and temporal resolution for core components of the water footprint), where the company did not have full visibility over broader aspects of the supply chain, primary data was supplemented with data from secondary sources which provided regionally specific data in accordance with level B. Similarly, the case studies in Chapters Four (pasta) and Five (tea) are based exclusively on data from secondary sources. However, they remain appropriate in this context because they have adopted an approach to data resolution which at a minimum complies with the requirements of level B and the need for regional or catchment specific data.

3.3.1.2 Guiding principles for the estimation of water footprints

Following the accepted methodology set out in *The Water Footprint Assessment Manual*, there are five main guiding principles involved in estimating a product water footprint (Hoekstra *et al.*, 2011). These principles – *supply chain boundary*, *water footprints directly associated with product inputs*, *overhead water footprints*, *distributing water footprints between products*, and *time* – were utilised in Chapter Six to conduct the original water footprint study on the potato crisp supply chain, and as such, are described below as they were applied by the author in this context. <u>However,</u>

it should be noted that these same principles were also considered by the authors of the secondary data case studies in Chapters Four and Five, and thus by detailing them here in the context of the potato crisp supply chain, they also illuminate and explain the terms and approach used in these chapters as well.

Principle 1 - Supply chain boundaries

WFA does not prescribe a firm rule for setting the supply chain boundary, and in particular, for truncating the analysis when going backwards along the supply chain. Other than a general instruction to include all processes within a production system which 'significantly' contribute to the overall water footprint (the definition of 'significant' being larger than either 1% or 5% of the total water footprint of the product), there is little guidance on what items should be included within the analysis (Hoekstra *et al.*, 2011, p. 10). However, in practice, a number of assumptions have been made in the water footprint community regarding the selection of appropriate *supply chain overhead* water footprint components (i.e. the water use associated with materials used in the factory for producing the product but not directly linked to its production) which are also adopted here (Ercin *et al.*, 2011 and Jefferies *et al.*, 2012). These assumptions involve taking account of only certain generic items within the supply chain overhead footprint as described in what follows (principle 4). As regards the *supply chain water footprint directly associated with inputs*, the approach in the water footprint literature, and again adopted here, has been to take account of all the items directly used in the manufacture of the product, but not necessarily to trace any tier two suppliers (i.e. they do not directly supply the product producer but rather they supply a tier one supplier that does). This is not to say that water associated with tier two suppliers is not impactful, but rather, that when, as here, the RQ focuses on water use at the product level, tier two suppliers have been universally excluded on the basis of significance (see, for example, Chapagain and Hoekstra, 2007; Chapagain and Orr, 2010; Aldaya and Hoekstra, 2010; Ercin *et al.*, 2011; Jefferies *et al.* 2012; Ruini *et al.*, 2013; Chico *et al.*, 2013). For example, in the context of agricultural supply chains, where the water used in crop cultivation often represents the overwhelming share of the total water footprint of the product (see Chapters Four, Five and Six), water use associated with any *inputs* at the agricultural stage (such as the *production* of fertilisers) has been excluded because by comparison it lacks significance (Chapagain and Orr, 2010; Chico *et al.*, 2013). Indeed, this focus on significance from a water perspective may well exclude some things which, from an LCA or carbon footprinting perspective, would be included. In particular, whilst the water footprint associated with transport and energy has been included in Chapter Six following the approach set out by Ercin *et al.* (2011), the tea and pasta case studies in Chapters Four and Five only partially account for these as neither category is particularly water intensive when compared to the total water used to produce a product (Hoekstra *et al.*, 2011, p. 11). The exception to this is if the source of energy in transport or energy production is biofuel, hydropower or biomass as all three are relatively water intensive (Gerbens-Leenes *et al.* 2009; Dominguez-Faus *et al.* 2009; Mekonnen and Hoekstra 2012).

Principle 2 - Water footprints directly associated with inputs - overview

The water footprint of a product is the sum of the water footprints of the process steps that occur within the supply chain boundary, either within direct operations or the broader supply chain. The total water footprint of an operational or supply chain process is given by the sum of the green, blue and grey water usage as shown in equation one below.[14,15]

$$WF_{proc} = WF_{proc,green} + WF_{proc,blue} + WF_{proc,grey} \quad [volume/time] \qquad (1)$$

Reflecting the definitions of blue and green consumptive use given in Chapter Two, equations two and three below set out the overarching formulas that guide the calculation of these components of the overall water footprint.

$$WF_{proc,blue} = \textit{Blue Water Evaporation} + \textit{Blue Water Incorporation} + \textit{Lost Return Flow} \; [volume/time] \qquad (2)$$

$$WF_{proc,green} = \textit{Green Water Evaporation} + \textit{Green Water Incorporation} \; [volume/time] \qquad (3)$$

Grey water is calculated by 'dividing the pollutant load (L, in mass/time) by the difference between the ambient water quality standard for that pollutant (the maximum acceptable concentration c_{max}, in mass/volume and its natural concentration in the

[14] Please note that all equations (1-10) included here have been taken directly from (Hoekstra *et al.*, 2011) and do not represent the authors formulations.
[15] Note that when calculating product water footprints, as opposed to their constituent processes, these are expressed as water volume per unit of product not time.

receiving body c_{nat}, in mass/volume)' (Hoekstra *et al.*, 2011, p.30). This is shown in equation four below:

$$WF_{proc,grey} = \frac{L}{C_{max} - C_{nat}} \qquad [volume/mass] \qquad (4)$$

Principle 2 - Water footprints directly associated with inputs – estimating the water footprint of agricultural crops

The principal input in the production of the crisp product in Chapter Six is the potato crop, the blue and green water footprints associated with which, were derived from the green and blue components of crop water use (CWU, m^3/ha) divided by the crop yield (Y, tonne/ha), as shown in equations five and six below (Hoekstra *et al.*, 2011, p. 41):

$$WF_{proc,green} = \frac{CWU_{green}}{Y} \qquad [volume/mass] \qquad (5)$$

$$WF_{proc,blue} = \frac{CWU_{blue}}{Y} \qquad [volume/mass] \qquad (6)$$

Primary data on crop potato yields was sourced directly from the farm in Chapter Six which grew the potatoes used in the manufacturing process. Crop Water Use was calculated with reference to the accumulation of evapotranspiration (ET, mm/day) over the growing cycle, as shown below:

$$CWU_{green} = 10X \sum_{d=1}^{lgp} ET_{green} \quad [volume/area] \qquad (7)$$

$$CWU_{blue} = 10X \sum_{d=1}^{lgp} ET_{blue} \quad [volume/area]$$ [16] $$(8)$$

The calculation spans the period from the day of planting (day 1) to the day of harvest (*lgp* stands for length of growing period in days) (Hoekstra *et al.*, 2011, p. 42). Evapotranspiration was estimated by using the *CROPWAT 8.0* model developed by the Food and Agriculture Organization of the United Nations (2015), comprehensive guides to which are provided by Allen *et al.* (1998), FAO (2008) and Hoekstra *et al.* (2011). Climate data for use in the model was sourced from the FAO *CLIMWAT* database (FAO, 2015a). Rainfall data for use in the model, covering the period 2006 to 2015, was sourced directly from the potato farm and adapted for use in *CROPWAT* using the process set out in FAO (2008). This involved estimating the rainfall associated with average, dry, wet and normal years. This was done in order to take account of temporal variations in

[16] The factor 10 is used to convert water depths in millimetres into water volumes (m^3/ha).

rainfall and ascertain what the normal level of rainfall is for use in the model. The standard potato profile that is built in to *CROPWAT* – which details crop parameters such as critical depletion fraction, yield response factors, rooting depth and crop height based on data from Allen *et al.* (1998) – was utilised. However, it was adapted where possible, using additional data from Allen *et al.* (1998) to reflect the growth stages of the potato crop type under analysis as detailed further in Chapter Six.

Calculating evapotranspiration using *CROPWAT* does not however include the water incorporated into the final harvested crop which also needs to be accounted for (see equations two and three). This was estimated with reference to the water fraction of the harvested potato crop which was supplied by the potato crisp manufacturer in Chapter Six. Note, the blue/green ratio of this incorporated water was assumed equal to the ratio of CWU_{green} to CWU_{blue} as suggested by Hoekstra *et al.* (2011).

The volumes of grey water were derived from the chemical application rate to the field per hectare (AR, kg/ha) times the leaching run-off fraction (α) divided by the maximum acceptable concentration (C_{max}, kg/m^3) and then divided by the crop yield (Y, tonne/ha) (Hoekstra *et al.*, 2011, p. 41). This is shown in equation nine below.[17]

$$WF_{proc,grey} = \frac{(\alpha \; X \; AR)/C_{max} - C_{nat}}{Y} \quad [volume/mass] \qquad (9)$$

As detailed in Chapter Six, for reasons of compatibility with sources of secondary data used in the analysis of the potato crisp case study, and in line with the majority of the water footprint literature, only nitrogen fertiliser was accounted for using primary data on application rates that was sourced directly from the potato farm that formed part of the case study. In addition, it was assumed that the leaching rate was 10%, natural nitrogen concentrations were zero and the maximum concentration in the receiving water body was 10 mg/l which accords with the USA's Environmental Protection Agency guidelines (see Mekonnen and Hoekstra, 2010a).

Where sufficient primary data was not available to calculate the green, blue and grey water footprints for crop inputs – principally in Chapter Six with reference to the sunflower oil and the production of potatoes in France – data was sourced from the *Water Stat* database (Mekonnen and Hoekstra, 2010a). This database, using very similar

[17] Note that it is only necessary to account for the pollutant which yields the highest water volume (Hoekstra *et al.*, 2011).

methods to those described above, provides details on the green, blue and grey water use associated with 350 crop and crop derived products, and does so for over 3,200 regions/provinces across the nations of the world thus complying with the spatiotemporal requirements of level B. It has received peer review endorsement both directly (see Mekonnen and Hoekstra, 2011 and Mekonnen and Hoekstra, 2010a), and indirectly in the large number of subsequent publications which have made use of the data which it contains (see for example Carr *et al.* 2012; Hoekstra and Mekonnen, 2012; Page *et al.* 2012; Ruini *et al.* 2013; Vanham and Bidoglio, 2013 and Ercin and Hoekstra, 2014).

Principle 2 - Water footprints directly associated with inputs – estimating the water footprint of industrial processes

When the process under analysis was industrial in nature as opposed to agricultural (i.e. when considering the processes within the factory which produces the potato crisps), it was not necessary to consider green water or evapotranspiration. In this case, data was collected from the potato crisp manufacturer on three categories of water use in its direct operations: 1) evaporative flow, 2) water volumes incorporated into products, and 3) return flows of water to catchments other than that from which the water was withdrawn 4) grey water discharges (Hoekstra *et al.*, 2011, p.69).

Principle 2 - Water footprints directly associated with inputs – estimating the water footprint of ancillary inputs used in production

For ancillary inputs into the production of potato crisps, mainly packaging inputs and salt, data was sourced from Ercin *et al.* (2011) and Ecoinvent (2013) respectively.

Principle 3 - Overhead water footprints

As mentioned previously, WFA does not provide clear instruction on how to treat supply chain overhead water footprints. Therefore, the approach here followed that adopted by Ercin *et al.* (2011) and Jefferies *et al.* (2012) who provide guidance on the appropriate selection of overhead water footprint components for analysis, together with the appropriate use of simplifying assumptions. These overhead components include: concrete, steel, paper, natural gas, electricity, steel and diesel.

Principle 4 - Distributing water footprints between products

In situations where a business produces more than one product, *overhead* water footprints have been distributed between these products according to product value. However, in the case where there was one input product and a number of output products, those water footprints that were described above as being directly associated with inputs (i.e. not overhead water footprints) have been distributed between end products according to what WFA describes as the *chain summation approach* (Hoekstra *et al.*, 2011). The formula for this is below:

$$WF_{prod}[P] = \left(WF_{proc}\,[p] + \sum_{i=1}^{y} \frac{WF_{prod}[i]}{f_p[p,i]}\right) X\, f_v[p] \qquad (10)$$

$WF_{prod}[P]$ is the water footprint of the output product p. WF_{prod} [i] is the water footprint of the input product i. WF_{proc} [p] is the process water footprint of the 'processing step that transforms the input products into the output products' (Hoekstra *et al.*, 2011, p. 50). The parameter $f_p[p,i]$ is the product fraction which is defined as the 'quantity of the output product obtained per quantity of input product' (Hoekstra *et al.*, 2011, p. 50). Parameter $f_v[p]$ is the value fraction which is defined as 'the ratio of the market value of…[the output] product to the aggregated market value of all the output products obtained from the input products' (Hoekstra *et al.*, 2011, p. 50).

By way of an example, if the water footprint of soya beans is 2,100 m^3/tonne, the product and value fractions related to soya bean oil produced that from soya beans are 0.18 and 0.34 respectively, and the process water footprint associated with producing soya bean oil is zero, then the water footprint of soya bean oil as a final product is 3,967 m^3/tonne i.e. (2,100/0.18) × 0.34 = 3,967 m^3/tonne.

Product and value fractions for potatoes and sunflower oil used in the production of potato crisps were calculated based in production data obtained from the potato crisp manufacturer.

Principle 5 - Period

Given the temporal variability in access to water, with fluctuations within and between years, and corresponding knock-on effects on demand, it is necessary to be clear about the period that any water footprint data refers to. For the factory stages in Chapter Six, all production data refers to financial year 2015 (January – December) as this was felt to reflect the most up to date depiction of on-site processes. However, the calculation of the water footprint of potatoes using *CROPWAT* was based on a normal year as mentioned, and likewise, the water footprint data from *Water Stat* is also based on annual average values during the period 1996-2005, thus taking account of this temporal variability.

3.3.1.3 Amendments to WFA methodology

Thus far the accepted WFA methodology has been detailed as applied in Chapter Six. However, given that water footprinting is a means to an end here (i.e. the valuation of water volumes) and not the end in itself, three minor modifications have been made to the methodology in order to ensure that the volumes arrived at are suitable for valuation purposes.

The first modification refers to the treatment of grey water. As mentioned previously, grey water is a *theoretical* volume that is defined as the amount of freshwater that is required to assimilate the load of pollutants given natural background concentrations and existing ambient water quality standards (Hoekstra *et al.*, 2011, p.2). Given this, it will be necessary to assume that there is not more pollution than assimilative capacity in the receiving water body in order to treat grey water as an actual, as opposed to theoretical, volume of water and one which is therefore suitable for valuation. Liu *et al.*, (2012) have examined historic 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 for, broadly concluding that excessive nitrogen and phosphorous discharges are more prevalent in the southern hemisphere, and that high general water pollution levels are to be found in tropical-subtropical areas. In the case studies that follow this chapter, where the supply chains encompass such geographies, the suitability of this assumption will be addressed.

The second modification refers to those aspects of the water footprint that are not geographically specific. These mainly include those items in the supply chain overhead footprint, and certain items that are used to produce the end product but which are sourced from a world market (e.g. the plastic in packaging inputs). Unless a geographical location is assumed for these items then it will not be possible to place a monetary value on the water that they represent. Therefore, it will be assumed that those non-geographically specific items are sourced from the country they are used in. For example, the water footprint associated with packaging inputs used in the production of potato crisps is assumed to occur in the same location as the factory which makes use of these inputs. However, given that these items are not sourced from a specific geography, they will never be a relevant change variable when comparing water values in different regions.

The final modification refers to the appropriate measures of water use. WFA measures water consumption, for reasons detailed in Chapter Two. However, as will become apparent in what follows, for some types of water use (e.g. domestic and industrial), it is water *withdrawal* that is the most common unit of measurement and thus the unit that monetary values are denominated in. Withdrawal values can be used to estimate the value of consumption. However, they would represent a lower bound value given that water consumption is referring to the most usefully applied component of water use and not a gross volume, some of which is not usefully applied given that it is simply returned from where it was withdrawn. Therefore, whilst the case studies in Chapters Four and Five did not report water withdrawn, the original case study conducted by the author in Chapter Six does make reference to water withdrawal as well as consumption in order to provide the fullest picture of the value of virtual water in a supply chain.

3.3.2 Phase two - Valuing product water footprints

Phase two introduces the wholly novel aspect of this research project given that, as mentioned earlier, previous attempts to chart virtual water (using whatever method) have neglected the robust measurement of full value. However, before discussion of which specific methods will be utilised to value virtual water, it is necessary to reflect upon the precise characteristics of grey and green water as this will influence the valuation approach adopted.

3.3.2.1 The nature of grey water

As shown in Figure 3.1 below, it is useful to view the water footprint of a single industrial production process in terms of input and output. What this shows is that green and blue water are the input, and that grey water (provided that it contains a pollutant load which is in excess of the pollutant load already in the receiving surface or groundwater body) or non-consumptive water use which is pollutant free (i.e. it is not consumptive because it is not incorporated into a product, it does not evaporate in production, and it is returned to the same catchment during the same time period), are the output. More specifically, it can be seen that gross blue water withdrawals register either as consumptive use and thus form the blue water footprint, or are emitted in the return flow as grey water or non-consumptive water. The key issues here are twofold. Firstly, there is no double counting between the grey and blue water footprints because the latter refers to consumptive water use only. Secondly, and more importantly, grey water, and any non-consumptive water, both stem from blue water and have no association with green water. In terms of grey water specifically, this is true whether the resulting pollution is assimilated adequately by the non-consumptive blue water in the return flow itself, or, whether it requires additional blue water in the receiving body to achieve this, where it is available i.e. where the grey water footprint is in excess of gross blue water withdrawals. In light of this then, because grey water ultimately originates from a blue water source, it will be argued in what follows that it is appropriate to value blue and grey water using the same methods i.e. to place the same value on blue water consumption and grey water degradation.

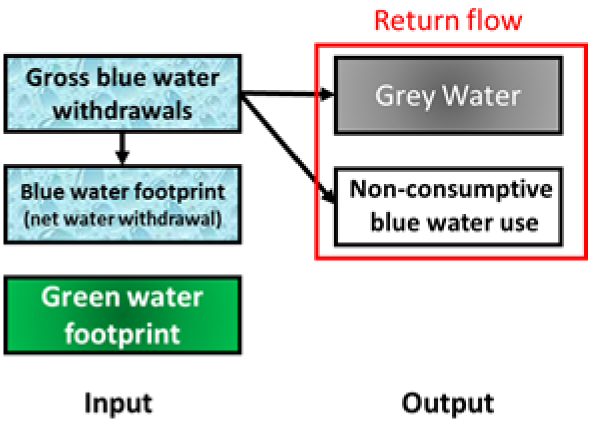

Figure 3.1 WFA as input and output (source: author).

3.3.2.2 The nature of green water

Green water has a lower opportunity cost, defined as the benefits foregone from possible alternative uses of the resource, when compared to blue water (Turner *et al.*, 2004, p.37). This arises from the fact that whilst green water can be used in agriculture and forestry, for example, in general blue water also has additional end uses, often with higher value added, such as industry. As a result, the valuation methods employed here will need to be sensitive to this issue.

3.3.2.3 Valuation framework

As mentioned earlier, the approach to the valuation phase reflects the level of data resolution that is being pursued here. What might be called 'headline' water footprint figures – which at a minimum will be compliant with what section 3.3.1.1 described as level B – will be relied on for valuation purposes i.e. the volumes of blue, green and grey water use. However, it should be remembered that what these figures do not tell the user on their own, and thus what will be beyond the scope of the valuation exercise, is catchment and sub-catchment specific details such as:

- Whether there is a spatial disconnect between places of blue water consumption and grey water pollution (as shown in Figure 3.2 below it is quite feasible that, even if the disconnect does not involve separate catchments, grey water may be discharged further down a watershed with different impacts).
- Detailed trade-offs, such as those between increased off-stream water use and the impact on in stream values.
- In depth assessment of variations in the timing of water availability which go beyond those suggested by the level of data resolution being pursued here.
- Thresholds beyond which the stocks and flows of ESS might be irreparably damaged.

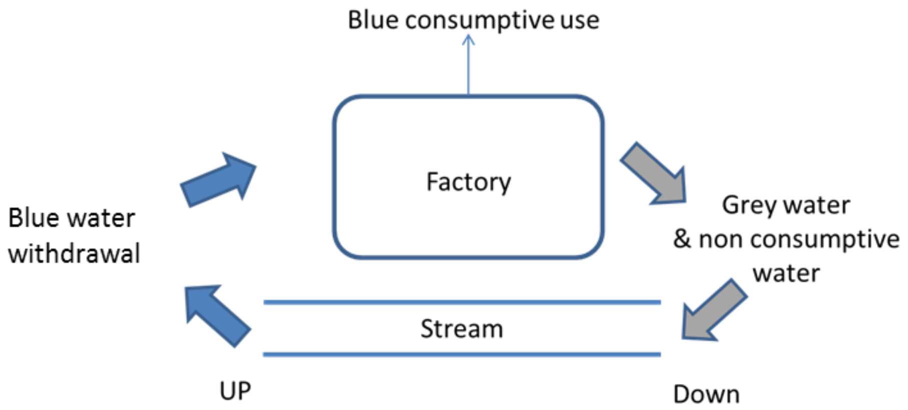

Figure 3.2 Spatial disconnect between water consumption and water pollution (source: author).

As a result, the aim of the valuation exercise (and this is reflected in the approach that is outlined below and in Part Three) is to provide an appreciation of the broad currents of monetary value associated with different types of water use in different areas, but not tied to a specific situation or scenario. In other words, the valuation approach is not looking to capture idiosyncrasies, but rather, what we might expect typical water resource values to be in a given location.

To enable this, the approach to monetary valuation adopted here is founded on BT together with an ESS framework. There are numerous categorisations of ESS, but perhaps the most widely referred to is that contained in the *Millennium Ecosystem Assessment* (2005). However, for the purposes of this paper, *The Common International Classification of Ecosystem Services*, or CICES, will be relied upon as this represents the state of the art in the field (Haines-Young and Potschin, 2013).[18] Table 3.2 below shows the CICES structure at the truncated three-digit level.

Table 3.2 CICES framework at the three-digit level

Section	Division	Group
Provisioning	Nutrition	Biomass Water
	Materials	Biomass, Fibre Water
	Energy	Biomass-based energy sources Mechanical energy
Regulation and maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota Mediation by ecosystems
	Mediation of flows	Mass flows Liquid flows Gaseous / air flows
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection Pest and disease control Soil formation and composition Water conditions Atmospheric composition and climate regulation
Cultural	Physical and intellectual interactions with ecosystems and land-/seascapes [environmental settings]	Physical and experiential interactions Intellectual and representational interactions
	Spiritual, symbolic and other interactions with ecosystems and land-/seascapes [environmental settings]	Spiritual and/or emblematic Other cultural outputs

Source: Haines-Young and Potschin, 2013.

3.3.2.4 Conceptualising the ecosystems impacted by green, blue and grey water use

In light of the ESS framework adopted, the key questions for monetary valuation are: 1) which of these services are impacted by the consumptive and degradative use of freshwater in the supply chain under examination, and 2) which of these services has a value to society. As noted previously, grey water, conceptually, is the volume of blue water required to abate thermal or chemical pollution. As such, a significant assumption made here will be to treat and therefore value blue and grey water in the same way, or

[18] In particular, the exclusion in CICES of the overarching category of supporting services, so as to avoid issues associated with double counting and thus ensure that ecosystem and economic accounts can be linked, is obviously crucial in this context.

in other words, assume that grey water pollution impacts on ESS in the same way that blue water consumption does. Whilst this may be true to a greater or lesser extent depending on the context, it is recognised that unlike blue water consumption which physically deprives other ESS of the associated volume of water, grey water may still be available for some ESS purposes even if in an impaired form. Moreover, unlike blue water which is consumed in the course of production, as mentioned in section 3.3.2.1 above, grey water may in fact not have been generated *during* the production process itself, but rather, registers because of the pollutants which are disposed of *afterwards*. Whilst these are fine points, the implication is that the value of blue water consumed, and grey water degraded, can both also be thought of as costs (or dis-benefits) i.e. the value the water could have been put to if it had not been consumed or degraded.

Table 3.3 below sets out the basket of ESS, accounted for at the group level (which CICES notes may be appropriate for accounting exercises such as this), which have been selected from the CICES classification as underpinning blue and grey water and thus which will be subject to valuation. Each of the ESS provides substantial value to society, and as a result, has been subjected to monetary valuation (see for example Turner *et al.* 2004). Of the six ESS selected, five are instream uses and one is off-stream use. The latter can be further sub-divided into agricultural, industrial and municipal uses.

Two things, in particular, should be noted about the ESS selected for analysis here. First, there are other ESS based values that could have been included in the valuation framework in addition to those noted in Table 3.3. For instance, in-stream values associated with hydropower and navigation. However, the decision has been made here to exclude these values because they do not represent activities which will be present in the majority of water bodies and rivers that the valuation method will be applied to. Similarly, the six ESS above, and particularly those which refer to the CICES ESS section Regulation and Maintenance, could encompass water values which are associated with wetlands. However, *functionally specific* values as they pertain to wetlands have been excluded here because, again, they represent idiosyncrasies which we do not know will be present in the various locations which the valuation approach is applied to. Second, by including off-stream uses, some of which will be subject to a market price, it is being assumed here, for reasons set out in Chapter Two, that the market price does not necessarily reflect the resources value in that use, never mind additional categories of value which that use will impact on. In other words, we are assuming that existing prices paid for water in a supply chain may or may not have sufficiently internalised the true value of water.

Table 3.3 Ecosystem Services underpinning blue and grey water

	ESS section	ESS group	In stream/off stream	TEV category	Market/non-market	Nature of demand
1	Provisioning	Surface or groundwater – non-drinking	Off-stream	Direct use	Market & non-market	Private good (intermediate input & final consumer good)
2	Regulation & maintenance	Waste assimilation	In-stream	Indirect use	Non-market	Public good characteristics (environmental service)
3	Regulation & maintenance	Hydrological – flood alleviation, and sediment retention	In-stream	Indirect use	Non-market	Public good characteristics (environmental service)
4	Regulation & maintenance	Wildlife habitat	In-stream	Indirect use	Non-market	Public good characteristics (environmental service)
5	Culture	Recreation	In-stream	Direct use (unpriced benefit)	Largely non-market	Public good characteristics (environmental service)
6	Culture	Other – existence, bequest, option	In-stream	Passive use	Non-market	Public good characteristics (environmental service)

Moving on to green water, it is quite clear from Table 3.2 that, as defined in the *Water Footprint Assessment Manual*, green water does not impact the same breadth of ESS quite so directly when compared to blue, and as conceived here, grey water. Therefore, the value of green water will be derived from a single ESS, crop production, located within the biomass grouping in the CICES classification above. More specifically, the contribution that green water makes in crop production will be estimated. It should be remembered here though that whilst green water gets confused with rainwater, it only refers to that portion of rain water that is evapotranspired by the crop (i.e. the portion that is usefully used). As such, the value of green water will be assumed to be synonymous with that portion of *artificially* applied irrigation water that is *consumed* by the crop, assuming that sufficient number of these values are identified in Part Two. If the aim had been to value rain water more broadly, as opposed to green water, then the values would potentially have been negative depending on the time of year as excess rain can lead, for instance, to water logging which impedes crop growth. However, this does not apply if the aim is simply to value evapotranspiration, the value of which, it is assumed here, does not vary depending on whether the water is artificially or naturally applied. In order to make this assumption however, it is also necessary to assume that the productivity of a unit of water which is evapotranspired is the same irrespective of timing (i.e. that there is a linear relationship between value and evapotranspiration levels) because *supplemental* irrigation would likely only be applied by the farmer to the extent necessary taking into account prior rainfall levels. Finally, by valuing green water as a single ESS, rather than the six which underpin blue water and grey water, the theoretical disparity in opportunity cost noted earlier is also reflected in the valuation framework.

It is worth mentioning her again that, as touched on in Chapter Two, a similar approach to valuing water use by companies in the supply chain has been advocated in the grey literature by environmental consultants Trucost (e.g. PUMA, 2010 and Danish Environmental Protection Agency, 2014a). However, in addition to not having received public peer reviewed endorsement, the approach by Trucost neglects the value of green and grey water and utilises a less encompassing basket of ESS in the valuation of blue water, omitting off-stream water use entirely.

3.3.2.5 Additionality of values, correspondence with TEV framework, and reporting values

As noted in Chapter Two, there is a correspondence between the various ESS and the TEV framework – as shown in Table 3.4 below – to the extent that by placing a value on the former an approximation of the latter can be derived.

Table 3.4 Ecosystem Services and the TEV framework

TEV category	Water related Ecosystem Services	
Direct use values	Provisioning: Food e.g. aquaculture. Fresh water for drinking and use in agriculture and industry.	Cultural: Bankside recreation. Boating. Fishing. Research and education.
Indirect use values	Regulation and maintenance: Waste assimilation. Flood alleviation.	Habitat support.
Passive use values	Cultural: Aesthetic Conservation	Symbolic Spiritual, sacred or religious

Source: Adapted from WBCSD, 2013.

However, it should be noted that off-stream water use in agriculture and industry (ESS 1 in Table 3.3) is not a final good but an intermediate input into production. As such it is subject to a derived demand (i.e. the demand is derived from the final good). Given this, in a strict sense, it does not make sense to apply the TEV framework to water use for these purposes which are private and rival goods i.e. the nature of the demand for them does not encompass the components of TEV. Nonetheless, the additionality of the various ESS values noted in Table 3.3 may still be appropriate depending on the configuration of a specific water basin. For example, it may be that the value for the off-stream use (for example in agriculture) could be added to the value of the in-stream ESS up to the point of diversion (although the agricultural value would need to an 'at source' value net of input costs, such as pumping costs, to make it commensurate with other in-stream values) (Figure 3.3). Indeed, as Brown (2004) sets out, because the in-stream ESS up until the point of diversion are non-consumptive in nature (i.e. no water is lost in use), it follows that the value of a cubic metre of water that is consumed in an off-stream use is the value of the *full* cubic metre in that use plus the value of the in-stream ESS up until that point.

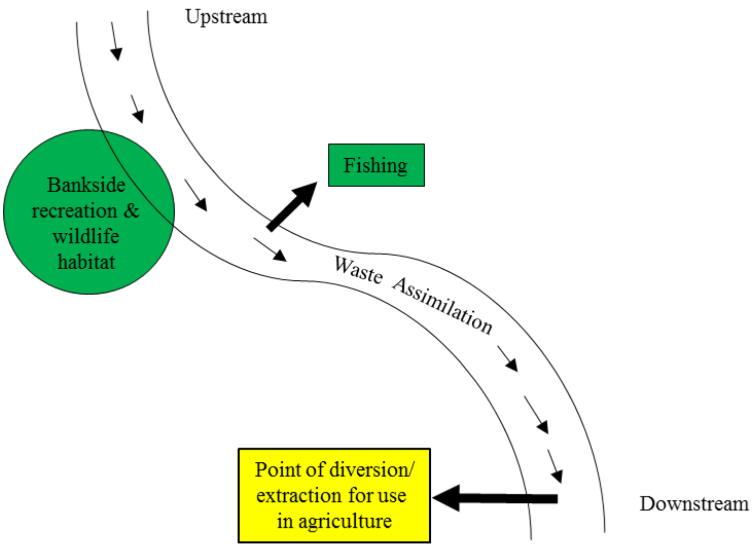

Figure 3.3 In-stream and off-stream values (source: author).

Since, as mentioned before (section 3.3.2.3), the analysis of value here is at a level of spatiotemporal detail that does not extend to a detailed analysis of each specific basin that individual supply chain stages span, the monetary values that are arrived at may need to be displayed separately for in-stream and off-stream uses given that there will not be a specific point of diversion. However, this will depend in part on the number and type of the values available in both categories. Even so, it is important to note that when full value or TEV is referred to in this context, as applied here, it is referring to the various ESS components and not suggesting that the nature of demand for water at any point along a supply chain encompasses all these components.

As shown in Figure 3.5 below, which shows a simple three stage supply chain that includes a consumer use phase, at each stage along the chain water values have two values for off-stream and in-stream water (the crop cultivation stage has an additional value to reflect the use of green water).

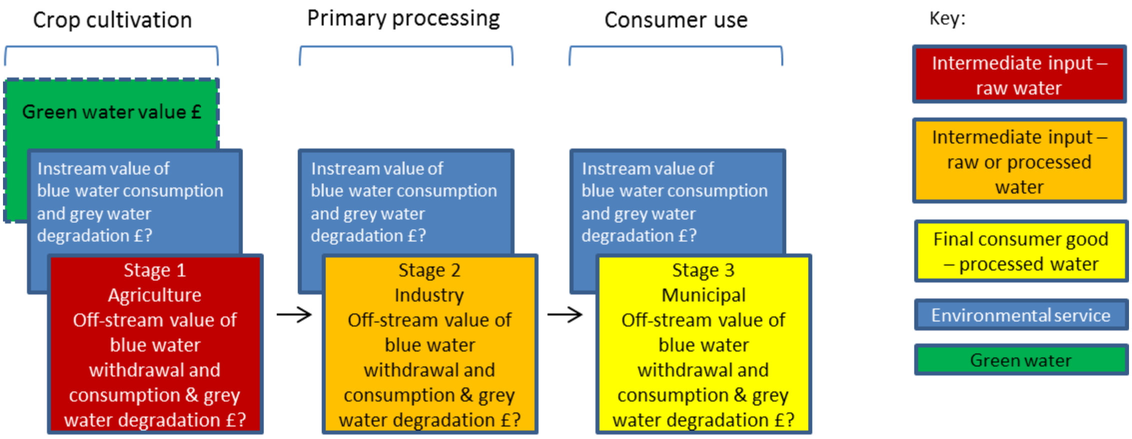

Figure 3.4. Supply chain water values encompassed by valuation framework (source: author).

Pre-empting somewhat the discussion in Parts Two and Three, Figure 3.4 also shows that the correct measure of water use differs by application. For example, agriculture water values are available which look at both withdrawal and consumption (thus enabling the approach to valuing green water noted earlier), whereas industrial water values are predominantly measured in in terms of withdrawal volumes. In addition, Figure 3.4 also makes clear that the values being assigned at each stage refer to differences in water quality from raw water use in agriculture to processed water in municipal uses. However, whilst quality is an important determinant of value, the value estimates in Part Two do not capture this dimension of water value directly. Indeed, the focus in Part Two is on water quantity, and how this varies by use and geography.

3.3.2.6 Benefit transfer

The approach to BT that will be adopted here will be set out once the first RQ has been addressed directly in Part Two. This will involve reviewing the number, geographical distribution and magnitude of the existing unit value estimates of water that correspond to the ESS framework adopted here, and assessing the ends to which these values can be put. However, before this commences, one overarching assumption will be necessary moving forward. This is that the quantity of water used at any point along the supply chain does not impact on its marginal value i.e. constant returns to scale. In other words, any values transferred will be on the assumption that the water used in the product supply chain does not impact the existing value drivers. If decision relevant values were required, this would need to be explored further with appropriate primary valuation techniques, but is in keeping with the level of spatiotemporal detail selected here and the idea that the method is a starting point not a definitive guide to supply chain water allocation and use.

Part Two

In light of the ESS framework adopted in Part One, Part Two now turns to a detailed analysis of the *unit* value estimates available in the literature which correspond to this framework. It should be noted here that this detailed review of the literature is another crucial aspect of difference between this research project and the approaches to the valuation of virtual water in the grey literature noted earlier. Indeed, as mentioned in Chapter Two, existing approaches appear to utilise constrained and/or unclear evidence bases. Therefore, Part Two aims to make apparent the existing knowledge base, and therefore, enable the discussion in Part Three about what approach this might support.

3.4 Compilation of the valuation literature

In compiling the literature for this analysis, a search of four specialist environmental valuation databases - EVRI, ValueBase SWE, Envalue, TEEB and the New Zealand Non-market Valuation Database - was conducted during the period April to October 2016. In addition, the reference sections of those papers identified were checked for additional relevant material. In all cases, the original papers identified in this search were consulted in order to obtain the original value estimates included here, the exception being where these were no longer available and thus a secondary reference had to be relied upon, provided one was available with sufficient detail. Nine publications, in particular, proved to be helpful in identifying relevant material (Young and Gray, 1973; Gibbons, 1987; Loomis, 1987; Colby, 1989; Brown, 1991; Frederick *et al.* 1996; Postel and Carpenter, 1997; Turner *et al.* 2004; Aylward *et al.* 2010). In the case of Gibbons (1987), Frederick *et al.* (1996) and Aylward *et al.* (2010), this arose because these studies had a similar aim, 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).

Studies were excluded where they were not published in English, where they referred to one-off unit value estimates for water but with little associated explanation regarding how this estimate was arrived at, where they used non-standard volumetric units of measurement (e.g. a bucket of water) and where 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. In addition, specifically with reference to agricultural water values, two further exclusion criteria were applied: 1) a small number of studies were excluded where they had taken a social accounting perspective which looked at what Bernardo *et al.* (1988) call *productivity related benefits* and which was inconsistent with the private accounting stance adopted across the other water use categories, 2) where the agricultural water value had been derived on the basis of a 'gross value' method – which simply divides the value of the crop by the water used to produce it – these values were also excluded as this method makes no attempt to estimate what *portion* of this value is attributable to water. As Young and Loomis (2014:90) state, this method 'implicitly assigns a zero shadow value to all purchased and owned inputs other than water,' thereby 'greatly overstating the correct welfare measure.'

In total, this process yielded 718 volumetric unit value estimates, across 126 sources which were undertaken between 1956 and 2015 (see Appendix 1 and 2 for the full list of sources). Reflecting what will become apparent is a broad division in the literature, the value estimates were divided into two groups:

- Those which refer to the USA – 409 estimates (or 57% of total estimates) from 69 sources.
- Those that refer to the Rest of the World (ROW) – 309 estimates (or 43% of total estimates) from 59 sources. Note that there were two sources which were common to both groups.

Figures 3.5 to 3.8 below set out the composition of the USA and ROW value estimates according to completion date of the source material, and the source type. As shown, unit estimation of water value is clearly more longstanding in the USA with 54% of the 409 USA estimates having been completed prior to 1987, compared to 5% of the 309 ROW estimates.

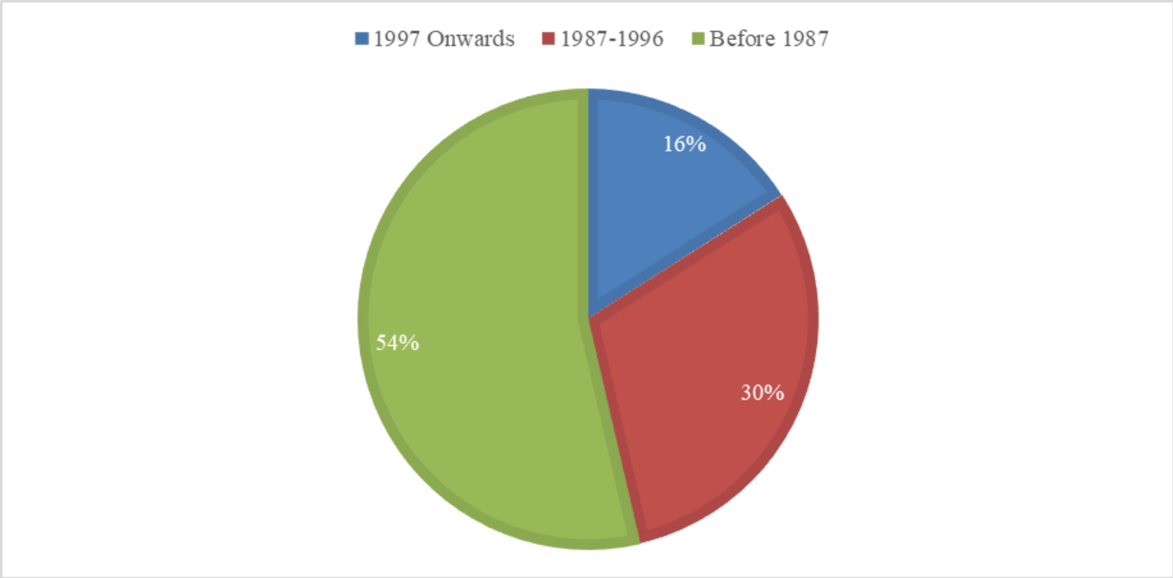

Figure 3.5 USA value composition by completion date of source material.

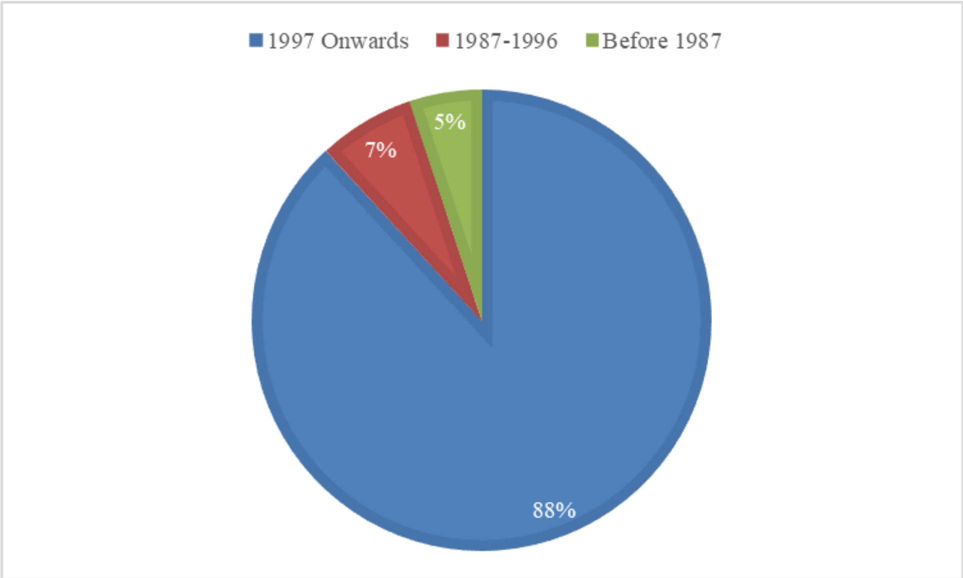

Figure 3.6 ROW value composition by completion date of source material.

Peer reviewed journal articles account for the largest share of source material (51% of USA sources and 63% of ROW sources) followed by reports commissioned or produced by governmental and academic research agencies (32% of USA sources and 22% of ROW sources). Table 3.5 below shows the specific journal titles which provide more than one source in either the USA or ROW groups.

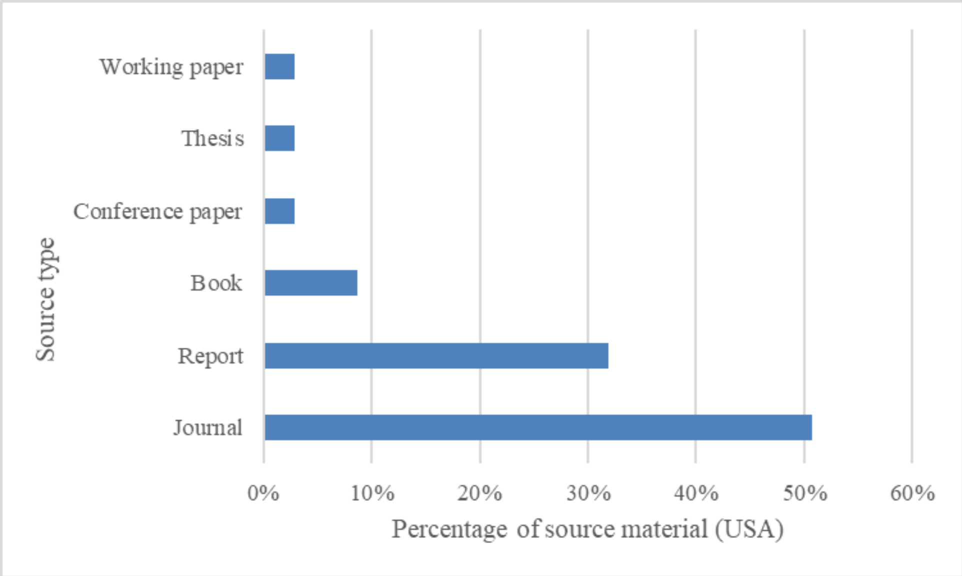

Figure 3.7 USA value composition by source type.

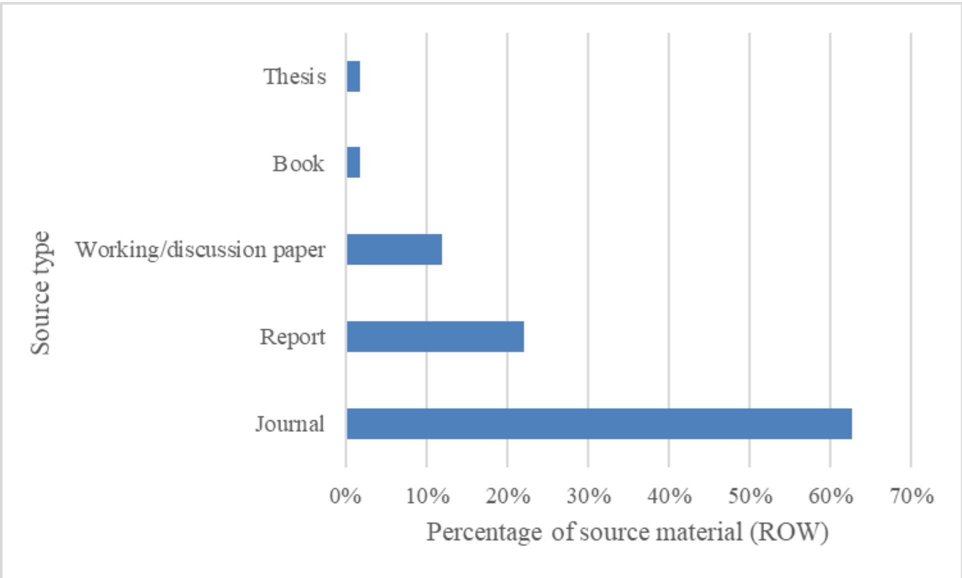

Figure 3.8 ROW value composition by source type.

Table 3.5 Journal sources in source material

USA journal sources	ROW journal sources
American Water Works Association	Agrekon
Land Economics	Canadian Water Resources Journal
Water Resources Bulletin	Ecological Economics
Water Resources Research	Journal of Environmental Management
	Science of the Total Environment
	Water Resources Management

3.5 Value standardisation

In line with the approach adopted by Frederick *et al.* (1996) – who attempted a similar exercise to this which required the updating of a large number of different water value estimates – as well as other authors (e.g. Rosenberger and Loomis, 2001), all value estimates have been temporally adjusted to 2014 US Dollars (USD) using the implicit price deflator (IPD) from the USA's Bureau of Economic Analysis (BEA) (see Appendix 3). Where the valuation year was not explicitly stated in the original study, the date of any underlying data used in the analysis was utilised as a proxy (where this was given as a range the last year was used), or if this was not possible, the year of publication. Where 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 using World Bank Purchasing Power Parity exchange rates applicable to appropriate valuation year, before being temporally adjusted using the IPD.

Where values were given as a simple range (e.g. \$10 – \$20) then 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.

Given that the majority of the 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. However, as will become clear in what follows, whilst the value estimates presented here have been recorded in acre feet, in the analysis of the supply chains in Chapters Four to Six, the value estimates will be 'deployed' in metric SI units i.e. cubic metres (1 AF = 1,233.48 m^3).

Many of the sources listed here, often for simplicity, report a value estimate as a single monetary figure rather than setting out any marginal relationship, even where one exists i.e. they are implicitly assuming constant returns to scale and an equivalence between marginal and average values. Where this abstraction has occurred, in Appendices 4 to 16 which set out each of the 718 value estimates, the single figure has been labelled as 'recorded.' However, where the source does provide a fuller picture of a marginal relationship in the form of multiple estimates (e.g. marginal recreation values with differing levels of water flow) then the median value in the range (and the range itself) has been included in order to ensure that this value is one which is observed. Any values included in this way have been labelled as 'summarised.' 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 taken into account (although as detailed below many of the most prominent parameters within each value category have been used to define the data). As a result, the assumption of constant returns to scale is implicitly being used, not just in the application of these values as mentioned previously, but also in the extraction of the value estimates from the literature.

For each value estimate, the measure of central tendency applied has also been recorded in Appendices 4 to 16. For example, if the value has been summarised then this will be the median value. However, in many CVM studies, for example, it is often the *mean* WTP that is reported as a single figure.

Finally, as reflected in Appendices 4 to 16, and as summarised in the forthcoming sections below, several sub-categories within each ESS have also been defined in order to properly delineate the respective data categories (Table 3.6).

Table 3.6 Sub-categories by type of use

ESS (type of use)	Sub-categories
Provisioning (Irrigation)	Per period, capitalised asset, on-site, in stream, short/long run, withdrawal, application, consumption, crop value (low & high).
Provisioning (Industry)	Sector
Provisioning (Municipal)	Domestic specific (Y/N)
Waste assimilation	Pollutant
Wildlife habitat	Per period, capitalised asset, wildlife type.
Recreation	Per period, capitalised asset, flow variation, recreation activity, site characteristics.

In order to classify the agricultural value estimates according to the sub-categories noted above, a number of assumptions were necessary for five specific valuation techniques as set out in Table 3.7 below. These assumptions were applied unless the source provided evidence to the contrary, and are based on the authors cited in Table 3.7, as well as the description of each technique provided in Chapter Two.

Classification of agricultural crops as either high or low value was based on El-Ahry and Gibbons (1988:14). It should be noted here that this classification, whilst referring to a generally applicable strata of crop values, came from the USA. Therefore, it was not sufficiently detailed to classify some crops grown in the ROW countries, and as a result, the summary values for high and low value crops grown outside of the USA that are presented in what follows should be treated with an element of caution. Where a crop was not classified for this reason, it has been labelled ‘not classified’ in Appendix Six.

Similarly, in Appendix Six, where a study was not specific about whether the crop was high or low value, or where this was unclear, the crop value is referred to as 'unknown.'

Table 3.7 Assumptions made in the classification of agricultural values

Technique	Assumption (unless stated otherwise)
Farm crop budget/ residual value	Volumetric measure is applied water (Gibbons, 1987; Naeser and Bennett, 1995). Values are short run and at site unless water procurement and fixed costs are explicitly subtracted.
HPM	Volumetric measure is withdrawn water. Values are long run and at site values (Loomis, 2014).
Linear Programming	Volumetric measure is applied water.
Water market transaction	Volumetric measure is withdrawn water. Values are short run and at site (Young and Gray, 1973; Young and Loomis, 2014).
Yield comparison	Volumetric measure is applied water. Values are Long run and at site (Young and Gray, 1973).

3.6 Nature of the value estimates

The 718 value estimates analysed here have been calculated using a variety of different market and non-market valuation methods. These include those cost based techniques, such as the alternative costs and avoided damages approaches, which are not based on the demand curve, as well as those stated and revealed preference techniques that give rise to genuine welfare estimates either in terms of Marshallian consumer surplus or the Hicksian compensating or equivalent measures. As a result, some of the estimates, chiefly across use categories, are not directly comparable in a strict sense. However, this is also true within categories, in particular for irrigation and recreational uses, which have seen the widest range of techniques utilised. In the case of the latter, value estimates for which are based either on the Marshallian consumer surplus or the Hicksian compensated demand function, given that the difference between these two welfare measures is due to the income effect and that expenditure on outdoor recreation represents a small share of income, there should not be a significant disparity (Rosenberger and Loomis, 2001). However, this caveat should be borne in mind and explains why, for each use category, value estimates will also be broken down by valuation technique, as well as by geography, where possible.

In addition to differences in welfare measures, 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 (see theoretical framework in Chapter Two). Indeed, in some cases it is not possible to identify what value conception is being identified as often the authors do not make this explicit. Considerations such as these will be considered in Chapters Four to Six when the values are applied to each case study.

3.7 Breakdown of values across ESS categories

In what follows, the breakdown of the 718 valuation estimates, across each of the six ESS categories in the valuation framework (see Table 3.3), will be assessed in turn.

3.7.1 Agricultural values

The value of water in agriculture here refers to the value of irrigation water that is artificially utilised in crop production. As discussed in Chapter Two, utilisation can be defined as the water that is withdrawn or diverted from a water source, that which is applied to the crop, or, that portion of applied water which is consumed during crop growth (sometimes referred to as net irrigation). The value of irrigation water can be further defined per period or as a capitalised asset, at the source of water extraction or at the site where it is used, in the long and short run, and finally, for different crop values. The number and composition of the agricultural values in the USA and ROW groups is addressed in turn below.

USA agricultural values

The search of the valuation literature revealed 210 *per period* agricultural value estimates, and 12 estimates of the capitalised value of agricultural water (see Appendix 4 and 5). This represents 29% and 2%, respectively, of the total number of value estimates and is thus the largest category of water use studied here.

Table 3.8 below sets out the mean, median, minimum and maximum *per period* values of agricultural water according to the sub-categories mentioned in Table 3.6 previously. As shown, the mean value for an acre foot of irrigation water in the USA, across all value estimates and sub-categories, is $105.30 (median value $65.30). In line with expectations, the at site value is significantly greater than the at source value (mean value 66% greater) given that the latter does not include water procurement costs. Likewise, the short run value for water is significantly greater than the long run value (mean value is 63% greater), and the value of water used in the production of high value crops is also larger than that used with low value crops (mean value 89% larger). However, whilst the value of irrigation water withdrawn is, as expected, lower than that which is applied, the evidence in Table 3.8 suggests that the value of water consumed, which should be the most valuable portion of irrigation water (Bernardo *et al.* 1988), has a lower value than the other volumetric measures. This should be treated with caution though given the imbalance in the relative number of valuation estimates across the volumetric measure categories, as well as the fact that the values for consumption all appear to have been associated with crops of low or unknown value.

Table 3.8 Agricultural water values (USA) by type

	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
All values estimates	105.30	65.30	-17.17	1,711.74	210
Location					
At site	113.63	74.80	-17.17	1,711.74	153
At source/in stream	68.63	52.73	0	213.35	32
Short/long run					
Short	110.75	95.51	4.33	407.56	66
Long	67.80	58.86	-17.17	247.69	86
Volumetric measure					
Withdrawal	45.54	21.78	4.33	197.16	18
Application	121.81	80.93	-17.17	1,711.74	147
Consumption	36.61	30.15	6.72	87.43	21
Crop value					
High	152.86	134.88	14.12	407.56	49
Low	112.63	65.90	-17.17	1,711.74	94

Note: Values in each sub-category have been calculated by holding other sub-categories constant. Number of estimates in sub-categories does not sum to 210 due to missing data. Negative values indicate that some crops are not viable in some locations.

Table 3.9 below breaks down the 210 estimates according to the principal valuation methods that were used in their estimation. Farm crop budgets (residual value) are the most popular method used in Table 3.9, likely owing to their relative simplicity when compared to the other techniques shown. Interestingly, four of the techniques have yielded mean and median water values in excessive of those attributable to the small sample of water market transactions in the data.

Table 3.9 Agricultural water values (USA) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
Production function	305.04	138.56	36.79	1,711.74	18
Farm crop budget	86.87	79.83	-17.17	247.69	70
Yield comparison	82.28	63.35	29.30	179.16	10
Linear programming	77.66	56.40	0	312.67	44
Water market	45.54	21.78	4.33	197.16	18
HPM	12.83	10.23	2.85	27.45	9

Figure 3.9 below depicts the geographical distribution of the source material associated with irrigation water values i.e. it shows how many times each state is specifically mentioned in the literature base (excludes papers which refer to broad geographies such as ‘western states). As can be seen, it is states in the south and east of the country which have received the by far the most attention to date.

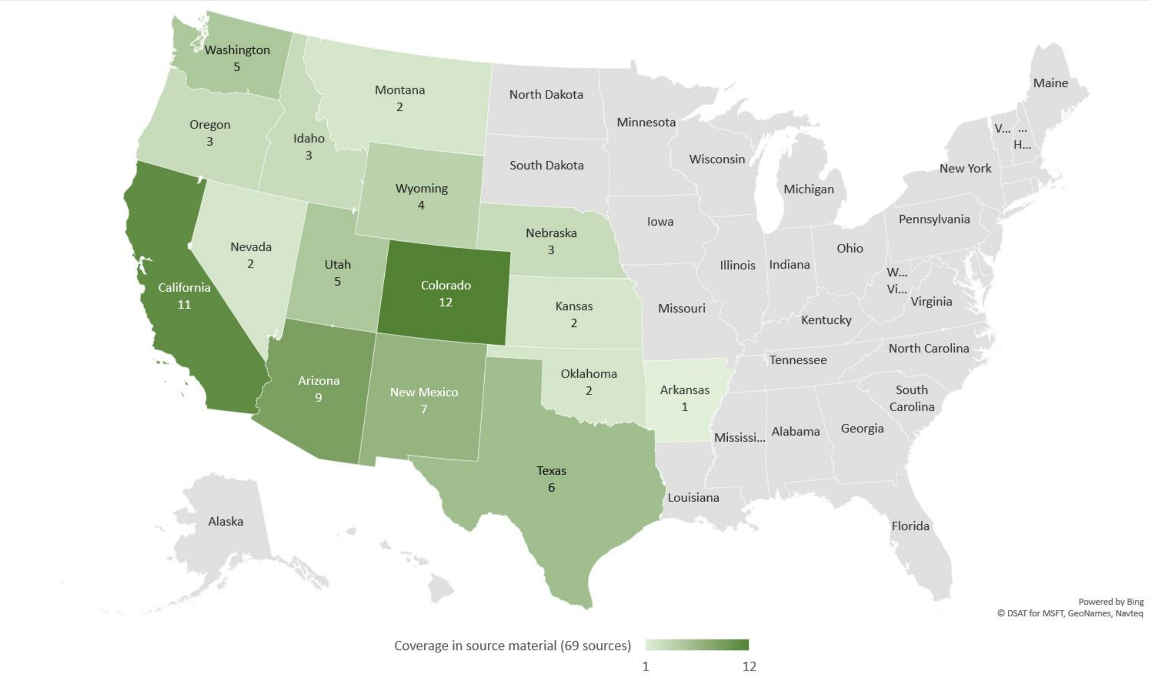

Figure 3.9. Coverage of agricultural water values (USA) in source material (used with permission from Microsoft).

In order to understand the geographical distribution of the value estimates as well as the source material, the 210 estimates were coded according to which USA Census Division they were located in. Table 3.10 below shows the mean, median, maximum and minimum values for each of the divisions for which data existed. Note there were no value estimates for the less arid eastern regions of the USA.

Table 3.10 Agricultural water values (USA) by census division

Census division	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
West North Central (4) [a]	69.25	63.57	5.07	176.27	9
West South Central (7) [b]	113.60	100.02	0	277.12	46
Mountain (8) [c]	93.26	58.82	-17.17	1,711.74	97
Pacific (9) [d]	135.70	73.72	8.87	956.42	36

[a] Iowa, Kansas, Minnesota, Missouri, Nebraska, North Dakota, South Dakota. [b] Arkansas, Louisiana, Oklahoma, Texas. [c] Arizona, Colorado, Idaho, New Mexico, Montana, Utah, Nevada, Wyoming. [d] Alaska, California, Hawaii, Oregon, Washington. Note: Number of estimates in sub-categories does not sum to 210 due to missing data.

As shown in Table 3.10, and Figures 3.10 and 3.11 below, the highest mean value ($113.60) occurs in the Pacific region, reflecting the preponderance of values in California. In terms of the mean value, this is greatest ($100.02) in the West South Central Division centred on Texas. However, these relative values should be treated with caution given that they are based on different numbers of estimates and differences in the composition of these estimates.

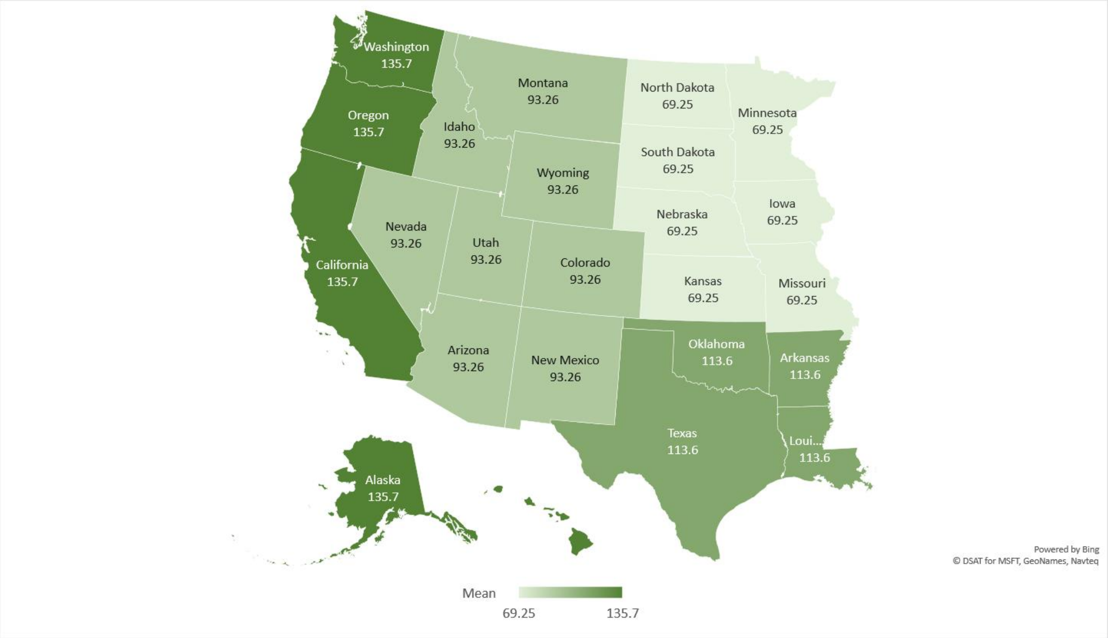

Figure 3.10 Mean agricultural value by Census Division (used with permission from Microsoft).

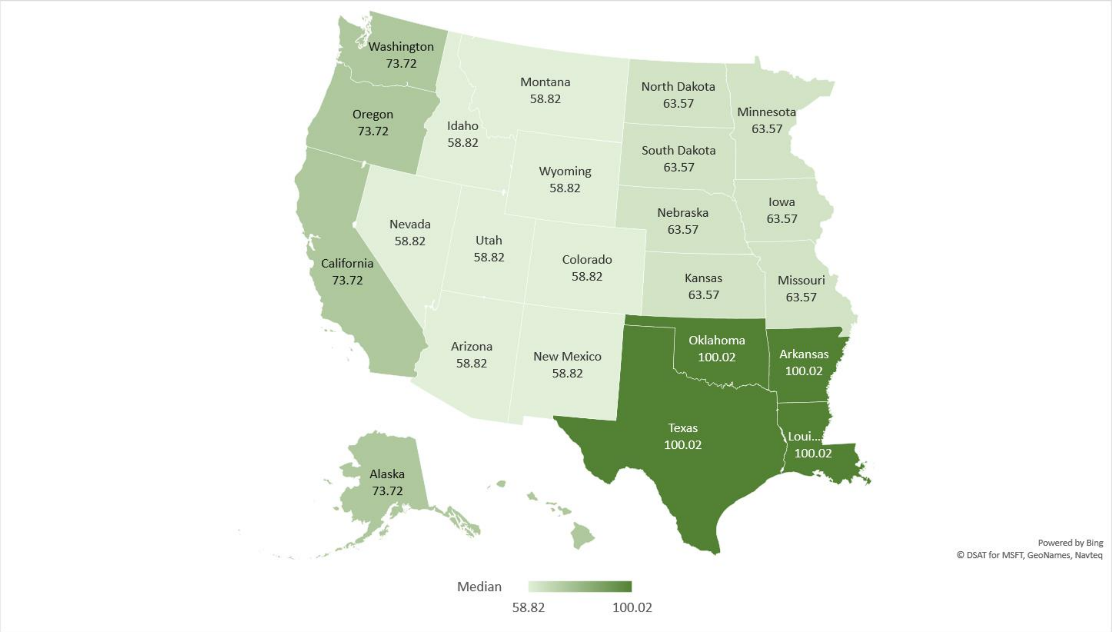

Figure 3.11 Median agricultural value by Census Division (used with permission from Microsoft).

Table 3.11 below summarises the 12 capitalised asset irrigation water values recorded. Young and Loomis (2014:38) point out that the most frequently used model for relating per period to capitalised values, typically produces capitalised values which are ten to twenty times larger than per period values. This ratio is in evidence here as the median capitalised value is 16 times greater than the per period value.

Table 3.11 Summary of agriculture values (USA) capitalised asset

Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
1,633.73	1,275.30	6,762.85	108.60	12

ROW agricultural values

The valuation literature yielded 145 estimates of the value of irrigation water outside the USA, across 21 countries and five continents (see Appendix 6). This represents 20% of the total number of value estimates and is therefore the second largest category of water use behind irrigation values in the USA (irrigation values as a whole make up approximately 50% of all values recorded).

The geographical distribution of irrigation ROW values is shown in Figure 3.12 and Table 3.12 below which sets out how many times each country was represented in the source material.

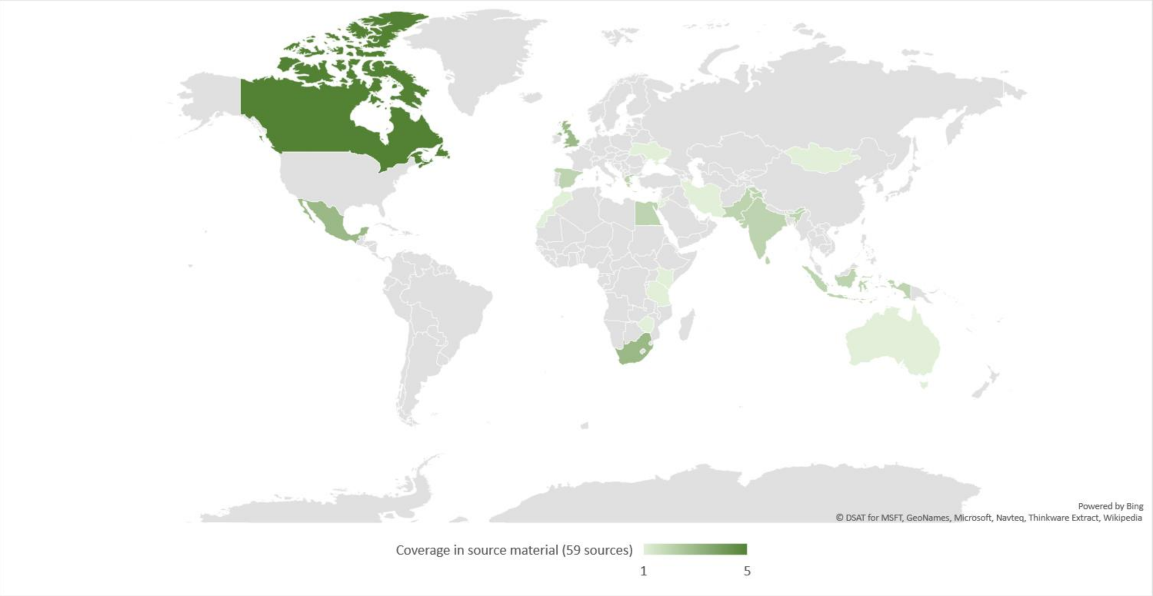

Figure 3.12 Coverage of agricultural water values (ROW) in source material (used with permission from Microsoft).

Table 3.12 Agricultural water values (ROW) by country

County	Continent	Coverage in source material (59 sources)
Australia	Australasia	1
Canada	North America	5
Greece	Europe	2
Cyprus	Europe	2
Egypt	Africa	2
India	Asia	2
Indonesia	Asia	2
Iran	Asia	1
Jordan	Asia	1
Kenya	Africa	1
Mexico	North America	3
Mongolia	Asia	1
Morocco	Africa	1
Pakistan	Asia	2
South Africa	Africa	3
Spain	Europe	2
Sri Lanka	Asia	2
Tanzania	Africa	1
Ukraine	Europe	1
United Kingdom	Europe	3
Zimbabwe	Africa	1

As shown in Figure 3.12 and Table 3.12, there is a significant dearth of irrigation values outside of North America. In particular, South America has no representation at all, and much of the other continents are only sparsely covered. This will obviously be a crucial factor in determining the approach to valuation that will be set out in Part Three given the agricultural based nature of each of the supply chains, and the fact they encompass geographies outside the USA.

Table 3.13 below sets out the mean, median, minimum and maximum *per period* agricultural values (ROW). Across the 145 value estimates, the mean irrigation water value is $550.32 which is significantly larger than the equivalent USA value. However, this is an average across 21 separate countries and is significantly impacted by a number of extreme values in individual locations (the largest ROW value recorded was $17,400 per AF which is 10 times greater than the largest USA value). As a result, the median value ($143.45) is a better representation of the value of water in agriculture in the ROW countries, but nonetheless, should still be treated with caution given the broad range of countries and regions that this encompasses. Again, as with USA irrigation values, the at site value is greater than the at source value (mean value 33% greater), and the short run value is greater than the long run value (mean value >300%), both as expected. However, in this case, the value for water consumed is also greater than that for water applied (mean value 11% greater) as we would suppose. As mentioned earlier, given the crop value classification used was not able to classify some of the crops grown in the ROW countries, the high and low crop value summary figures should be treated with a degree of caution. Nonetheless, as expected, the mean and median values are both higher for high value crops when compared to low valued crops.

Table 3.13 Agricultural water values (ROW) by type

	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
All values estimates	550.32	143.45	0	17,441.36	145
Location					
At site	299.86	180.94	0	1,846.55	53
At source/in stream	225.82	144.85	0	902.50	18
Short/long run					
Short	182.26	83.56	0	995.94	47
Long	60.25	37.94	11.71	146.24	11
Volumetric measure					
Withdrawal	124.62	103.92	15.48	337.22	7
Application	525.02	148.44	5.79	7,450.84	68
Consumption	581.58	318.21	61.91	2,141.89	12
Crop value					
High	2,644.70	905.50	14.79	17,441.36	13
Low	471.87	173.70	0	7,450.84	66

Note: Number of estimates in sub-categories does not sum to 145 due to missing data.

Table 3.14 below sets out the mean and median irrigation water values – across the sub-categories – by continent. Again, it is extremely difficult to compare across regions given disparities in the number and make-up of the value estimates in each location. The resulting mean and median figures should therefore be treated with caution and as only broadly indicative of any geographical variation in the value of irrigation water across the continents covered.

Table 3.14 Agricultural water values (ROW) by continent

Continent	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
North America (exc USA)	266.56	180.94	0	1,648.71	48
South America	N/A	N/A	N/A	N/A	0
Europe	1,573.48	653.15	55.28	7,450.84	18
Africa	167.74	81.48	0	902.50	44
Asia	920.04	98.36	15.84	17,441.36	34
Australasia	16.92	16.92	16.92	16.92	1

Table 3.15 below breaks down the 145 value estimates according the principal methods that were used in their estimation.

Table 3.15 Agricultural water values (ROW) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
Yield comparison	833.22	77.34	11.71	7,450.84	31
Opportunity cost	392.06	392.06	392.06	392.06	1
Production function	385.85	225.82	5.79	1,648.71	24
Farm crop budget/residual value	366.69	141.36	0	3,271.40	49
Linear programming	309.50	146.24	21.68	905.50	7
Stated preference (CVM & DCE)	150.43	12.54	0	871.54	13
HPM	143.45	143.45	143.45	143.45	1
Benefit Transfer	33.55	30.48	0	70.18	3

As with the agricultural values presented previously, the disparity between the mean and median values above suggests that a number of extreme values are skewing the summary measures. Excluding the opportunity cost approach, which only accounts for one value estimate, the production function approach has produced the highest median value, which is in keeping with the analysis of USA agricultural values presented in Table 3.8. However, perhaps reflecting the more contemporary nature of the ROW value estimates, SP techniques are present for the first time (although the farm crop budget remains the most popular technique used).

3.7.2 Industrial values

Industrial values arise when water, which may be self-supplied, is used in industry for the purposes of, for example, cooling, the processing of raw materials, and general overhead requirements in factories such as cleaning and hygiene. Values are driven, predominantly, by the type of industrial use that the water is put to, and the water quality requirements associated with this. For example, water for food processing usually must meet stringent quality standards, which are unlikely to be necessary for water used for cooling.

USA industrial values

The search of the valuation literature yielded 42 standardised estimates (6% of total estimates), from 10 sources, which are summarised in Table 3.16 below and set out in full in Appendix 7. These estimates span sectors such as textiles, food, mining and minerals, chemicals, paper, metals and power generation.

Table 3.16 Industrial water values (USA) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
Added value	299,999.97	16,022.07	420.33	2,214,103.49	14
Residual imputation	1,529.48	1,529.48	1,529.48	1,529.48	1
Cost of intake	290.86	169.35	48.22	1,049.36	9
Alternative cost	173.75	21.31	8.80	1,414.90	18

Compared to the previous section on agricultural values, what is most noticeable with the industrial values (USA) is that they are far few fewer in number, and they have been estimated using several early approaches, many of which, are no longer considered valid. Furthermore, the 42 industrial values recorded here are essentially the same ones that were previously recorded by Young and Gray (1973), and Gibbons (1987), as the estimation of industrial water value does not appear to have evolved in the USA in the intervening years since these sources were published. Indeed, the critique of these values also remains the same as that noted at length in these two sources, a precis of which will be covered below as a means of properly contextualising the values noted in Table 3.16 and differentiating those values that provide more realistic and reliable estimates.

The first method mentioned in Table 3.16, *added value*, involves 'the estimation of the ratio of some measure of value added (as of income to primary resources) to a measure of water utilized (sic)' (Young and Gray, 1973, p. 162). The measure of value can include direct value added but also indirect value added such as regional multiplier effects. The main criticism of the added value approach is that 'it does not reflect the productivity of water in the process' (Ibid, p.164). That is, an industrial water user which has a high added value but which makes use of limited quantities of water, will have a high added value per unit. Because of this, and the inclusion of some indirect effects (sources 43 and 48 in Appendix 7 include some indirect effects), the values noted in Table 3.16 above for the added value approach are very high and do not bear comparison with values from other techniques. Indeed, they could have been excluded here on this basis in a similar manner to the exclusion of what were described as 'gross values' in agriculture (section 3.4). However, they have not been excluded because there are very few industrial values in existence, and their inclusion highlights the methodological advancement which has seen more realistic value estimates produced.

The *residual imputation* method, which is the same basic approach to valuation as that used in farm crop budgets, has also faced criticism when applied in an industrial setting. Gibbons (1987:49) makes the point that this approach is ‘unreliable when water costs are a miniscule element of total costs,’ as is often the case in industry. Similarly, values derived from the *cost of intake* approach – which equates the cost of water intake with its value in the production process – are described by Young and Gray (1973:162) as ‘only…indicative of what industries can pay for water, and hence, …of limited value for private or public water allocation decisions.’

By contrast, the *alternative costs* approach is the method which is overtly preferred by Gibbons (1987) and seemingly advocated by its use, and comparative lack of criticism, in Young and Gray (1973). The alternative cost approach equates value with the internal costs of water recirculation. That is, industry ‘should be willing to pay only up to what it would cost to produce water of adequate quality through treatment and reuse (Gibbons, 1987, p.49/50). In this context, the alternative cost estimates represent the only category which provides a reasonable estimate of the value of water to industry in the USA. However, as will be demonstrated in the next sub-section on industrial values (ROW), in more recent years, methods for estimating industrial values have expanded to include effective approaches in addition to alternative costs, but this has only occurred in settings outside the USA.

Figure 3.13 below shows the geographical distribution of the source material pertaining to industrial values in the USA i.e. the number of instances whereby each state is referred to in the literature. Unlike agricultural values, there is a distinct focus on the central portion of the USA, likely reflecting the historical location of much industrial enterprise.

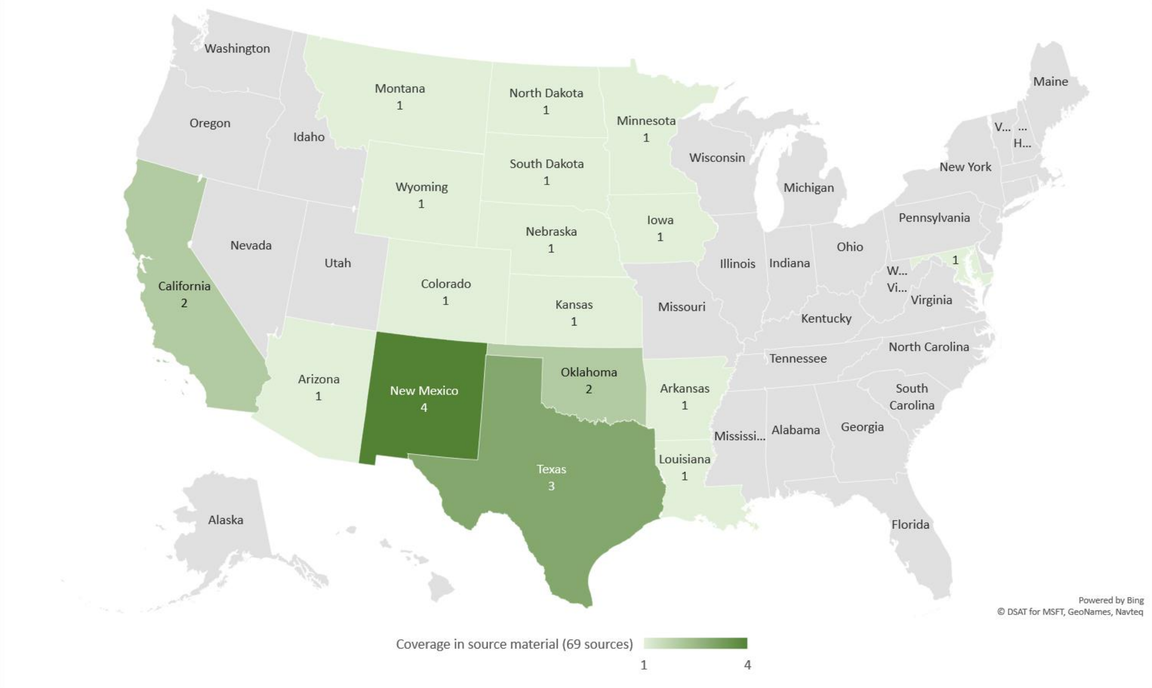

Figure 3.13 Coverage of industrial water values (USA) in source material (used with permission from Microsoft).

ROW industrial values

The detailed literature search returned 89 estimates of the value of industrial water outside of the USA (see Appendix 8), across six countries in North America and Asia. This represents 12% of the total number of estimates recorded. The geographical distribution of industrial (ROW) estimates is detailed in Figure 3.14 and Table 3.17 below which sets out how many times each country was represented in the source material.

To an even greater extent than with agricultural (ROW) values, industrial values are clearly only available for a handful of countries. However, importantly, these values encompass a wide range of sectors (e.g. food and beverages, textiles, chemicals, paper, metals, mining and minerals, pharmaceuticals and power generation) and water uses, which as noted earlier, are the primary drivers of the value of water in industry.

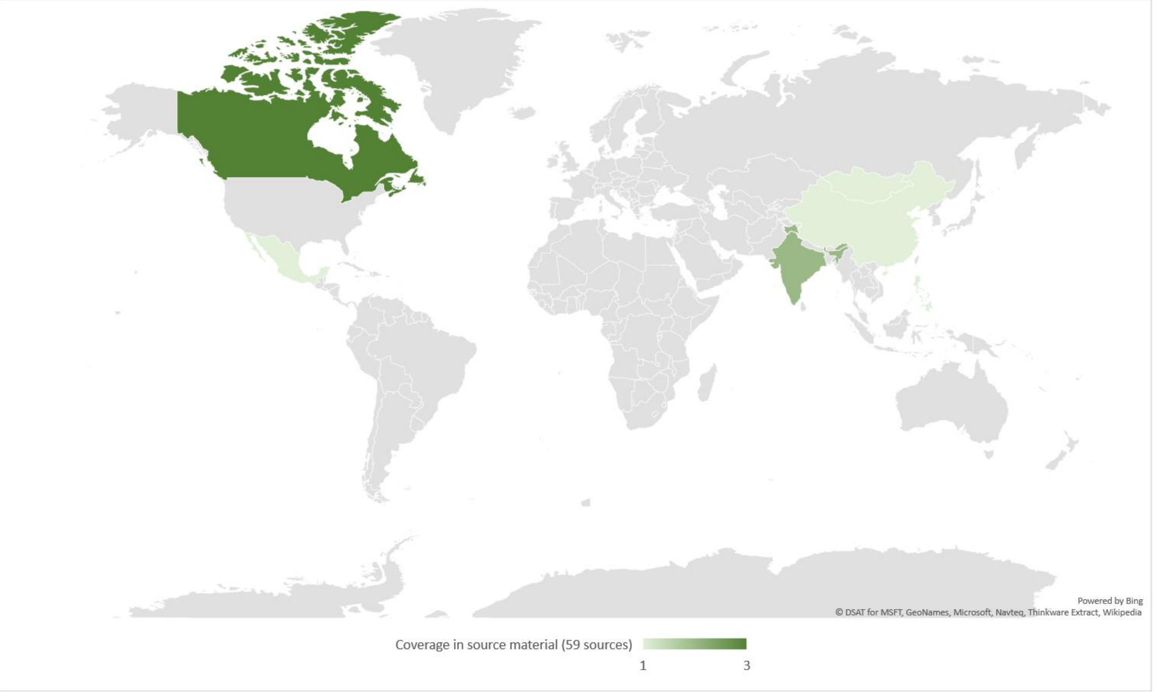

Figure 3.14 Coverage of industrial water values (ROW) in source material (used with permission from Microsoft).

Table 3.17 Industrial water values (ROW) by country

County	Continent	Coverage in source material (59 sources)
Canada	North America	3
China	Asia	1
India	Asia	2
Mongolia	Asia	1
Philippines	Asia	1
Mexico	North America	1

Table 3.18 below sets out the mean, median, minimum and maximum *per period* values of industrial (ROW) water. There are a small number of added value estimates (four) in evidence in Table 3.18 which once again are significantly inflated when compared to the values form alternative approaches. As a result, these estimates can be ignored on the same basis as that set out above in relation to industrial (USA) values. However, the estimates provided by the production function, input distance function and cost function approaches come from three papers (Wang and Lall, 2002; Kumar, 2004; Renzetti and Dupont, 2002), each of which represents significant methodological advancement in the field of industrial water values.

Table 3.18 Industrial water values (ROW) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
Added value	17,991.65	18,499.16	4,425.83	30,542.43	4
Production function	6,699.52	4,142.42	44.78	24,030.50	16
Alternative cost	1,927.84	623.88	53.56	18,542.59	43
Benefit transfer	1,914.67	1,914.67	1,914.67	1,914.67	1
Opportunity cost	1,056.17	1,056.17	1,056.17	1,056.17	1
Input distance function	1,045.57	519.79	190.73	5,016.40	9
Cost function	80.05	43.99	2.20	452.44	14

Note: Number of estimates in sub-categories does not sum to 89 due to missing data.

Wang and Lall (2002) have undertaken what Gibbons (1987) described at the time as a 'vein hope,' referring to the use of a production function to provide statistical estimates of the productivity of a unit of water in industry. Based on an aggregate data set of 2,000 medium and large state owned factories located in China across 16 economic sectors, their marginal productivity approach treats water as one input to the production function along with labour, capital and raw materials. Wang and Lall's results have been criticised by Renzetti and Dupont (2002:3/4) who suggest that their regression equation suffers from simultaneity bias and the presence of uncorrected multicollinearity. Nonetheless, Wang and Lall's approach is the only one that exists that looks to estimate the physical productivity of a unit of water in industry. In response to their criticisms of Wang and Lall's approach, Renzetti and Dupont (2002) have developed a cost function based approach which estimates the shadow value of water in 14 industries in Canada. More specifically, their cost function approach characterises 'the firm's short-run or restricted technology and then estimates the reduction in short-run costs that follow from providing the firm with an incremental increase in its intake water' (Renzetti and Dupont, 2002, p.17). The resulting cost savings thus represent an estimate of the firms marginal WTP for that water in a short run context. It should be noted that Renzetti and Dupont point out that their estimates are reflective of the relatively low level of regulation in Canada at the time, which perhaps explains the comparatively low values noted in Table 3.18 above. A similar approach to Renzetti and Dupont (2002) has been that proposed by Kumar (2004) who utilise an input distance function, which is the dual of the cost function, to estimate the shadow value of water in nine industries in India. Based on sales, input costs and water consumption from a survey of 92 companies, this approach, the author argues, is preferable to cost and production functions of Renzetti and Dupont (2002) and Wang and Lall (2002) because it allows the possibility of multiple outputs, and does not require information on input prices. In addition, the input distance function does not require the assumption of cost minimisation by firms.

In addition to these three papers, in a report commissioned by Natural Resources Canada, Environment Canada and the University of Saskatchewan, Bruneau (2007) has made use of a more traditional *alternative cost* approach to estimate the average value of water in 14 industries in Canada. This source is worthy of particular note here because the 43 shadow value estimates that it derives (see Table 3.18 above) are for process water, raw water, as well as consumed water.

3.7.3 Municipal values

The water used for municipal and domestic purposes refers to that which is used around the home, both indoors (e.g. for cooking, washing and hygiene) and outdoors (e.g. lawn sprinklers), and that used in commercial (non-industrial) business activities.

USA municipal values

The search of the valuation literature revealed 25 *per period* municipal value estimates and 16 estimates of the capitalised value of municipal water (see Appendix 9 and 10). This represents 3% and 2%, respectively, of the total number of value estimates.

Table 3.19 below sets out the mean, median, minimum and maximum *per period* values of municipal (USA) water.

Table 3.19 Municipal water values (USA) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
All value estimates	230.83	91.96	4.36	739.92	25
Demand function	434.48	530.35	66.21	739.92	11
Water market	70.82	44.33	4.36	214.78	14

As shown in Table 3.19, over half the value estimates refer to water market transactions where municipal authorities have leased water, predominantly from agricultural rights holders. The remaining values have all come from simplified demand functions which, in overview, have produced significantly higher values than those noted in the market transactions. However, the values derived by Gibbons (1987) and Young and Gray (1973) – who between them account for approximately half the demand function values – have both been estimated using what appears to be a standard formula for the integral of a demand function, which Young and Loomis (2014) and Young and Gray (1973) both ascribe to James and Lee (1971). The most accessible version of the formula – provided by Young and Loomis (2014:238) – is set out below:

$$V = [(P\ x\ Q_1^{\frac{1}{E}}) \,/\, (1 - \frac{1}{E})] \ast [(Q_1^{1-\frac{1}{E}}) - (Q_2^{1-\frac{1}{e}})]$$

Where:

E = Elasticity

P = Price

Q = Quantity

The application of the formula necessitates four data points:

1) An initial price observation (P_1);
2) An estimate of water usage (Q_1)
3) The change in quantity that is the subject of valuation ($Q_2 – Q_1$); and
4) A price elasticity of demand (this is assumed to be constant between Q1 and Q2 and not equal to 1.0).

With these, it is possible to estimate a second point on the demand function, from the initial price and quantity observation, the total area under which, represents the value of treated water that is delivered to the home. However, if the assumption is made that municipal water is priced to fully recover the costs of supplying it (i.e. there is no producer surplus), the ‘average revenue can be subtracted from the total WTP to derive net consumer surplus’ which reflects the value of raw water at its source (area A in Figure 2.1) (Young and Loomis, 2014, p.239). This is shown in the equation below (Young and Loomis, 2014, p.239):

$$CS = V - [(P_1)(Q_1 - Q_2)]$$

Figure 3.15 below shows the number of times that each state within the USA was mentioned in the literature. As with agriculture, there has been a very distinct focus on the south and west of the country to date.

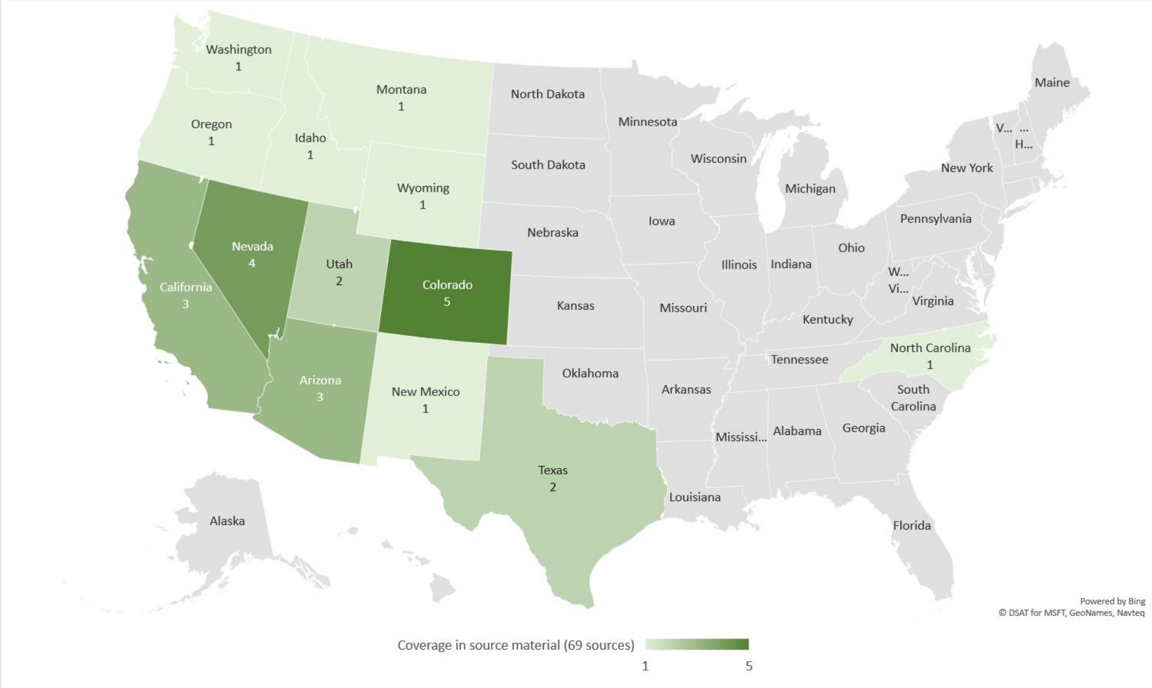

Figure 3.15 Coverage of municipal water values (USA) in source material (used with permission from Microsoft).

ROW municipal values

The valuation literature provided 65 estimates of the value of municipal water outside the USA (see Appendix 11), across 13 countries in North America, Asia and Africa. This represents 9% of the total number of value estimates. The geographical distribution of municipal (ROW) values is detailed in Figure 3.16 and Table 3.20 below which sets out how many times each country was represented in the source material.

Whilst the number of municipal (ROW) values is greater than the number of municipal (USA) values, as with the agriculture and industry, these values are clearly not evenly distributed and only encompass a handful of, mainly developing, countries.

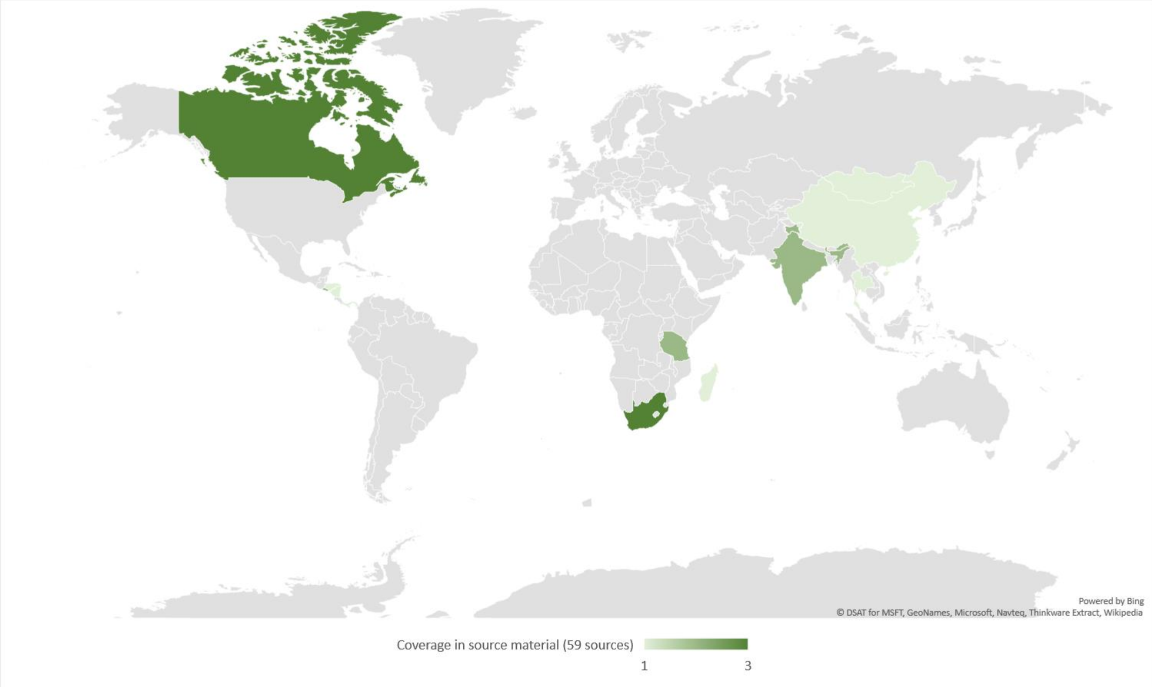

Figure 3.16 Coverage of municipal water values (ROW) in source material (used with permission from Microsoft).

Table 3.20 Municipal water values (ROW) by country

County	Continent	Coverage in source material (59 sources)
Canada	North America	3
China	Asia	1
El Salvador	North America	2
Honduras	North America	1
India	Asia	2
Madagascar	Asia	1
Mongolia	Asia	1
Nicaragua	North America	1
Palestinian Territory	Asia	1
Panama	North America	1
South Africa	Africa	3
Tanzania	Africa	2
Thailand	Asia	1

Table 3.21 below sets out the mean, median, minimum and maximum *per period* values of municipal (ROW) water. As shown, there have clearly been a wider range of techniques applied in the estimation of municipal (ROW) values when compared to municipal (USA) values. These techniques include more modern approaches, in particular SP techniques. In addition, the value ranges associated with the municipal (ROW) values are also clearly much larger than the municipal (USA) values. This likely reflects the fact that some of the values in Table 3.21 – for example the largest value of $22,959 – are comparatively small scale water vendor transactions for subsistence

purposes, WTP for which, may be significantly greater than when the water is for municipal purposes more generally.

Table 3.21 Municipal water values (ROW) by method

Method	Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
All value estimates	1,700.05	482.83	49.05	22,959	65
Stated preference	2,667.56	965.63	163.93	21,419.74	18
Price	1,841.67	409.82	114.75	22,959	25
Demand function	1,170.16	433.12	49.05	4,736.83	8
Benefit transfer	815.08	815.08	64.20	1,565.95	2
Opportunity cost	419.44	258.23	132.78	867.32	3
RP	416.85	311.47	229.50	803.26	7

Note: Number of estimates in sub-categories does not sum to 65 due to missing data.

3.7.4 Waste assimilation values

The value of water for waste assimilation stems from the potential of rivers and streams to dilute wastes and thus decrease any damages that may be suffered by other water users and also reduce the costs associated with waste treatment. Waste assimilation values are dependent upon the specific pollutant, ambient water quality standards, and the level of water flow.

The detailed literature search discovered 13 standardised value estimates for waste assimilation (2% of the 719 estimates collected), stemming from six different sources, which are detailed in full in Appendix 12 and summarised in Table 3.22 below. All of these estimates are for waste assimilation values in the USA only. For 12 of the 13 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.

Table 3.22 Summary of waste assimilation values (USA)

Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
7.53	2.05	0.23	39.24	13

What is most noticeable about the waste assimilation value estimates is that the majority come from just two sources – (Meritt and Mar, 1969 and Gray and Young, 1974) – which appear to be the only examples which have been specifically focused on the estimation of dilution values. Moreover, whilst both are now somewhat dated, they have not been improved upon. 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. Likewise, Gray and Young (1974) is the only paper cited at any length by Gibbons (1987) in their review of a similar nature. Young and Loomis (2014:277) attribute this lack of interest in waste assimilation in recent years to the fact that the primary conclusion which came out of these early studies was that there were cheaper options for ameliorating pollution damages than dilution. Similarly, Gibbons (1987:64) concludes that 'as the external costs of water quality degradation are increasingly charged to polluters, more process changes [i.e. waste treatment] will become cost effective, and the demand for waste dilution will continue to decrease.' This lack of contemporary interest in dilution values is also confirmed by the fact that whilst several studies were discovered on the unit value of waste water treatment (see Appendix 13 Waste Water Treatment Plants WWTP) – studies which were all less than ten years old – they did not look to consider the amortization of this value over a given quantity of dilution water. Overall, the value of water for dilution purposes appears to be relatively low according to Table 3.22, and it has arguably become lower in light of improved waste treatment technology and improved water quality standards in many parts of the world.

Figure 3.17 below shows the states within the USA which the six waste assimilation sources cover. Once again, it is the western states which predominate. Note: each state in the USA is represented in Figure 3.17 below because Gray and Young (1974) report waste assimilation values for 22 major river basins which span the lower 48 states of the continental USA.

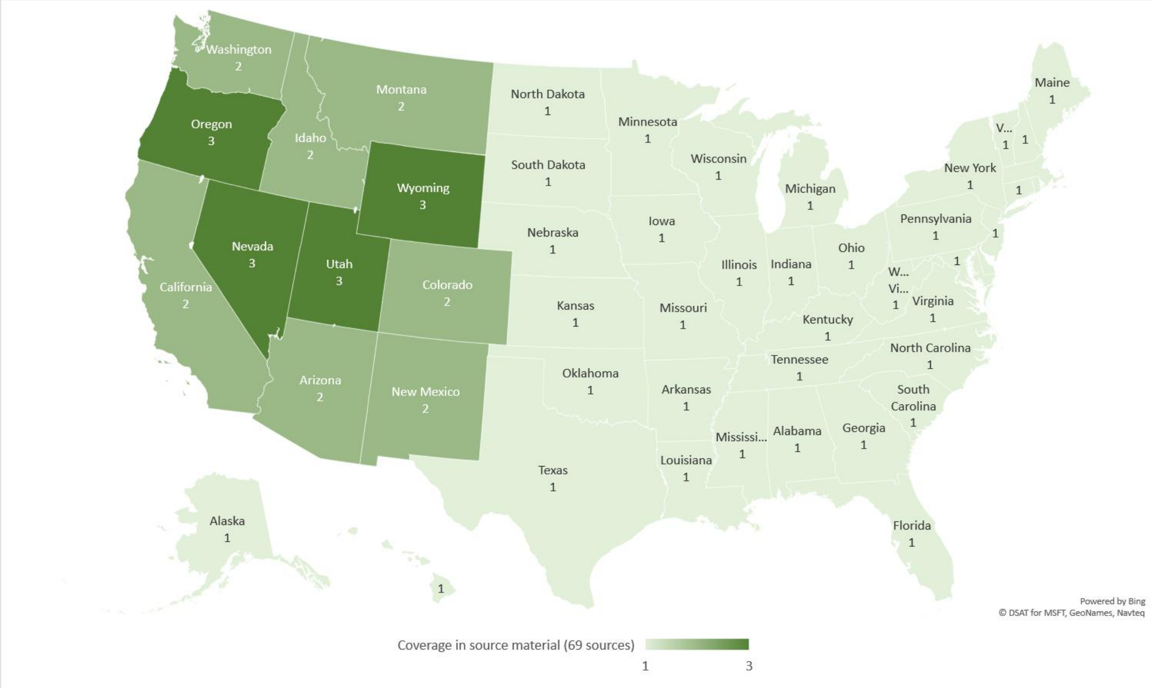

Figure 3.17 Coverage of waste assimilation water values (USA) in source material (used with permission from Microsoft).

3.7.5 Wildlife habitat values

Wildlife habitat refers to the role that water plays in terms of providing a habitat for fish and other, potentially endangered, species. Whilst it is possible to view the values of recreation activities such as waterfowl hunting, fishing and angling as proxies for wildlife habitat because they capture part of this value, in this context, values for wildlife habitat have been taken from studies which isolate the value of water for this purpose. This has been achieved either by focusing on the volumes and values of water that have been specifically provided, via a market transaction, for augmenting low flows for wildlife habitat, or by focusing on commercial activities (such as commercial fishing) where the proxy value does not include a non-commercial or recreational element. The detailed literature search discovered 24 per period value estimates, originating from seven sources, and 18 capitalised asset values originating from four sources, which met these criteria. These estimates, which are all for USA, are detailed in full in Appendix 14 and 15 and summarised in Table 3.23 and 3.24 below. Note, whilst *functionally specific* values of wetlands have been excluded in this context, two estimates reflecting the value of water for wildlife habitat in a wetlands setting have nonetheless been included here because they are reflective of the more generic value of water for this purpose.

Table 3.23 Summary of wildlife habitat values (per period) (USA)

Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
59.67	55.61	0.16	161.08	24

Table 3.24 Summary of wildlife habitat values (capitalised asset) (USA)

Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
1,375.43	1,240.31	57.09	5,369.49	18

The capitalised asset value is 23 times larger than the per period value, again in line with Young and Loomis (2014:38).

Figure 3.18 below shows the states within the USA which the wildlife habitat values cover. Once again, the west and south of the country are the only areas with any representation.

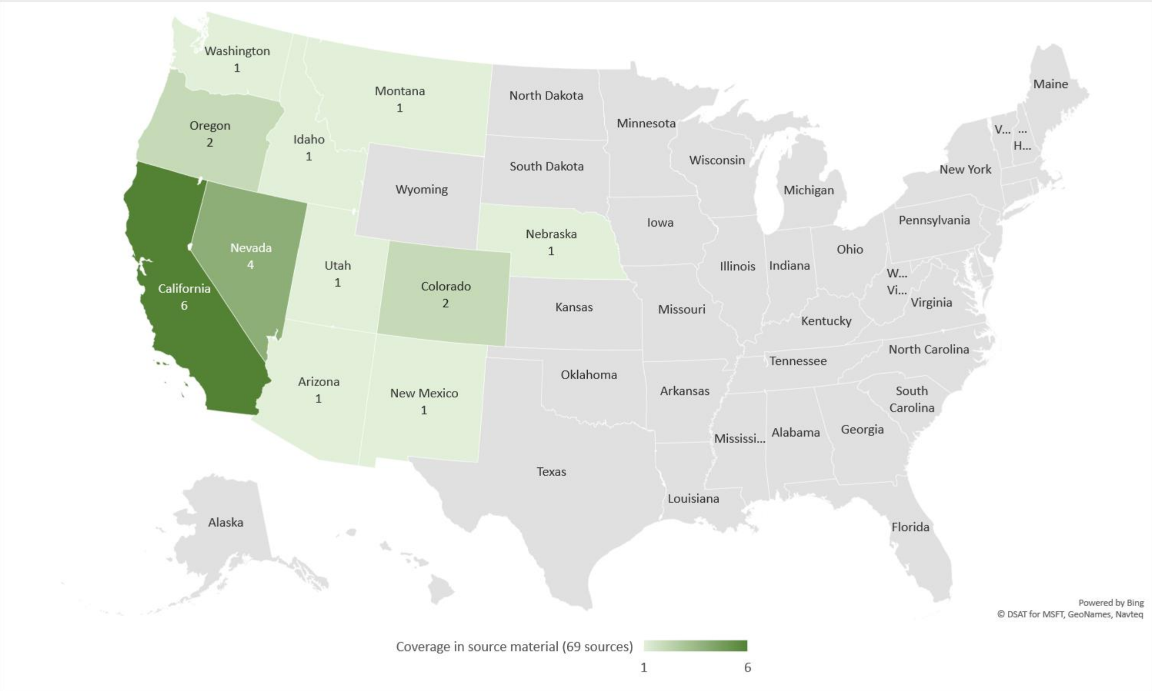

Figure 3.18 Coverage of wildlife habitat water values (USA) in source material (used with permission from Microsoft).

3.7.6 Recreation values

The detailed literature review discovered 49 standardised estimates (7% of total estimates) of the recreational value of water stemming from 27 separate sources (see Appendix 16). These estimates include the recreational benefit provided by direct access to water in the form of rafting, kayaking and fishing, as well as shoreline based activities such as waterfowl hunting, camping and hiking which are enriched by proximity to water. The recreational value estimates, which again are all originate from the USA, are summarised in Table 3.25 below. Note, as was the case for wildlife habitat, one study has been included here on the recreation value of water in a wetlands setting. Again, however, this is not a functionally specific value and is reflective of the more generic value of water for this purpose.

Table 3.25 Summary of recreation values (USA)

Mean value (2014 USD/AF)	Median value (2014 USD/AF)	Minimum value (2014 USD/AF)	Maximum value (2014 USD/AF)	Number of estimates
43.57	13.32	0.33	550.12	49

The majority of the 27 estimates are for river based recreation. They have been estimated using CVM and TCM approaches which specifically look to establish the relationship between variation in the level of flow in a river and the associated marginal value (see column three in Appendix 16). As Gibbons (1987) suggests, this is in contrast to early attempts to establish the unit value of water for recreation which often began with a recreational value for a site, in dollars per day, and then looked to amortise this over a specific quantity of water. The problem with this, as Gibbons suggests, is that it produces average not marginal values, and more importantly, the denominator is difficult to define. Indeed, four of the estimates here appear to have used such an approach, the denominator for instance being different fill levels in a reservoir, but Gibbons also notes instances where the surface area of a water body has been used.

Figure 3.19 below shows the states within the USA which the recreation estimates cover. As with the other categories of water use, values are concentrated in the south and west of the country, in this case, with a particular emphasis on Colorado.

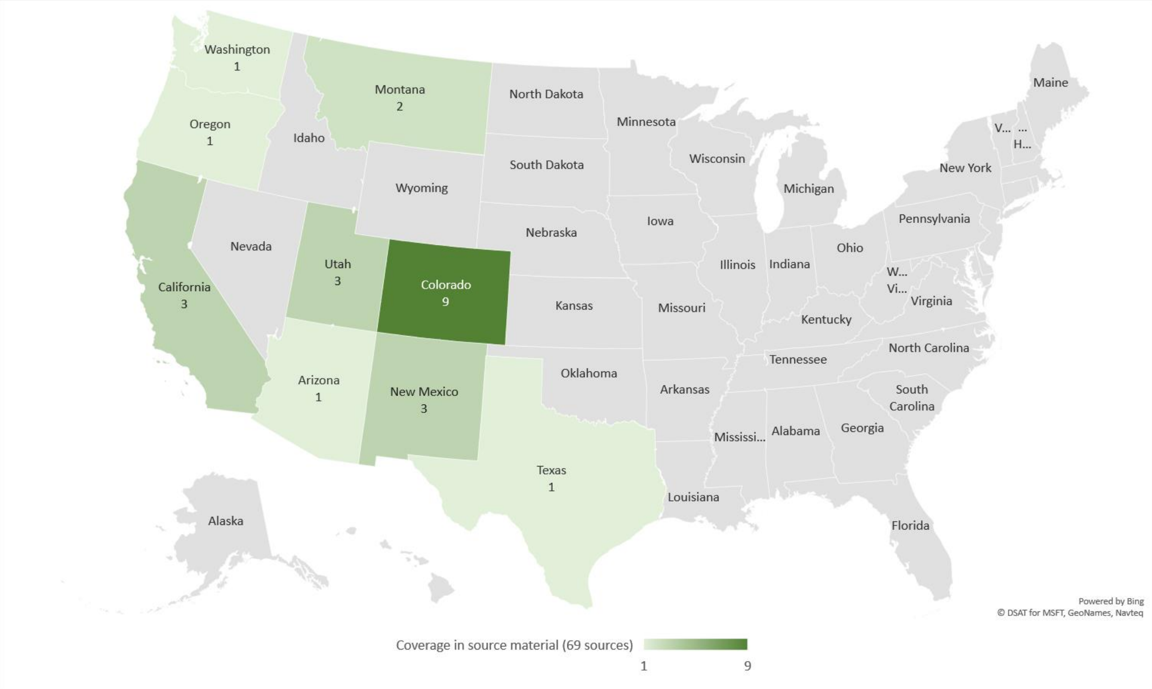

Figure 3.19 Coverage of recreation water values (USA) in source material (used with permission from Microsoft).

3.7.7 Hydrological and passive use values

The compilation of the valuation literature described above did not uncover any suitable hydrological or passive use values for use in this context. In the case of the former, a limited number of values are available for the hydrological services that are performed by wetlands. However, no unit values were available for the hydrological services provided by rivers and water bodies more generally. Given, as mentioned, the method that is being developed here is deliberately not looking to capture any idiosyncrasies at the locations the valuation approach is applied to, and as such is not including functional values which are specific to wetlands, hydrological values have necessarily been excluded altogether as the hydrological services provided in the two contexts are very different.

Similarly, there was only one study on passive use values which was denominated in unit value terms (Loomis, 2012). What is more, however, these values are highly location specific, and in a strict sense, as mentioned, they are only appropriately derived in situations where water is an end consumer good. As a result, passive use values have necessarily been excluded in this context as well.

3.8. Summary (Part Two)

In summary, Figures 3.20 and 3.21 below set out the number of studies, and value estimates, applicable to the ESS categories for which values were found during the detailed literature search which was described in section 3.4.

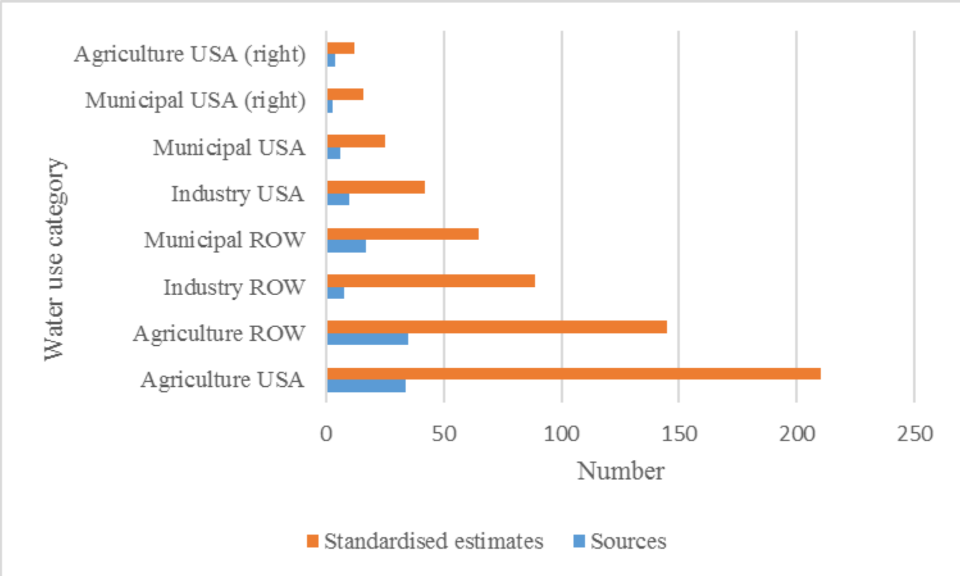

Figure 3.20 Off-stream water values (number of standardised estimates and sources per category).

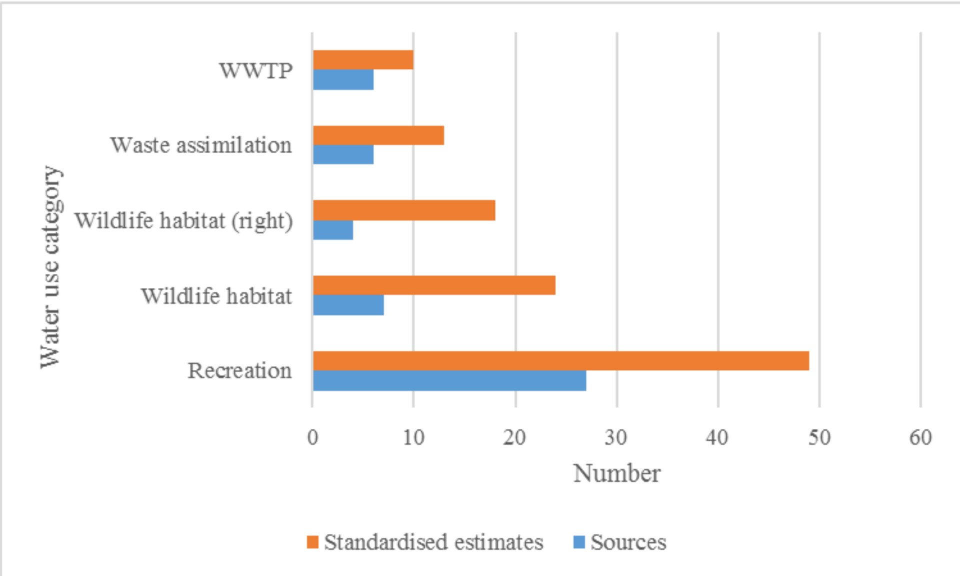

Figure 3.21 In stream water values (number of standardised estimates and sources per category).

Agricultural values dominate the off-stream water values accounting for approximately 50% of total estimates. Similarly, recreation values are the preponderant in in-stream values, although they are far less significant as a percentage of total estimates (7%) when compared to agricultural values. Nonetheless, behind agriculture, recreation is the second most studied area in the literature with 27 sources reporting recreational values, compared to 69 for agriculture as a whole.

Figure 3.22 below sets out the median values for each category assessed. The median value has been chosen here in order to lessen the impact of large outlying values which would impact the mean value. In addition, as mentioned in section 3.6.2, owing to the obsolete nature of some of the valuation techniques that were used to estimate industrial values, the median USA and ROW values reflect only the more relevant valuation techniques that have been applied.

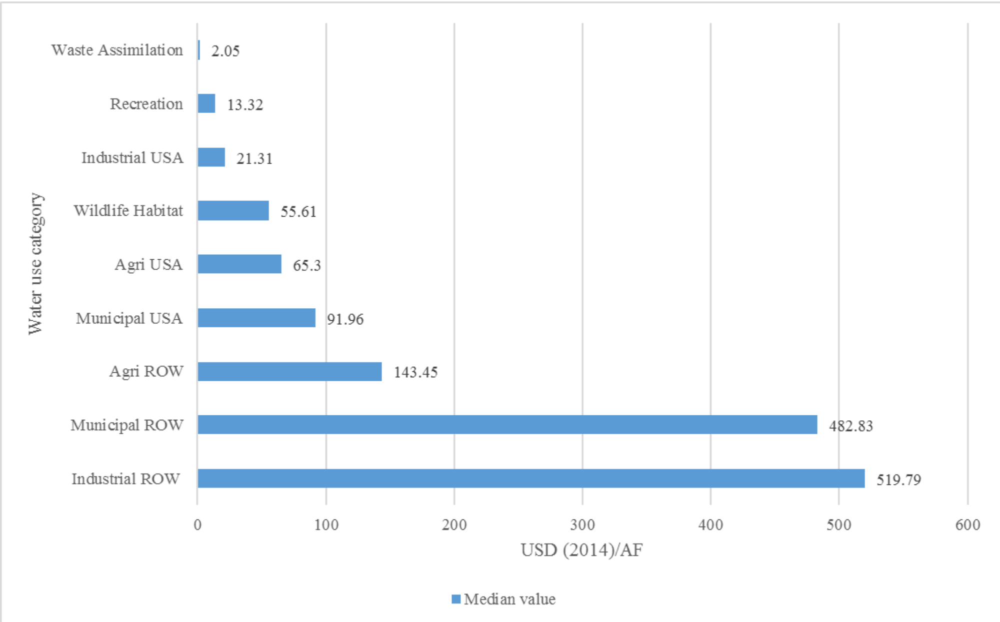

Figure 3.22 Median unit values of water across categories
(Note: median value for industry ROW is for input distance function, cost function and production function techniques only; median value for industry USA if for alternative cost techniques only).

The pattern shown in Figure 3.22 is similar to that noted by Briscoe (1996, p.182) in their assessment of the relative value of water, albeit just in the context of the western USA. However, in this instance, water for wildlife habitat and recreation, which might be considered synonymous with what Briscoe (1996) labelled ‘environmental purposes,’ exhibits a lower value than that for agriculture.

Overall, what is clear from the analysis above is that the 718 value estimates are very unevenly distributed both across water use categories, and geographies. Indeed, the arid regions of the western USA alone account for approximately 57% of all values. Similarly, agriculture (across ROW and USA) accounts for 51% of all estimates. In addition, it is also quite clear that, particularly outside the western USA, the summary mean and median value figures quoted above for each ESS category obscure a large amount of variation, be that in terms of the volumetric measure or crop value in agriculture, or the validity of the valuation techniques that have been applied in industry. Furthermore, the absence of any passive use values, denominated in unit value terms, means that the estimation of full value or TEV in what follows can only ever be synonymous with direct and indirect use values and thus a partial estimate. Finally, the meagre number of agricultural values measured in terms of water consumption calls into question the method for valuing green water set out in Part One. As a result, the approach to valuing water in the supply chain, which will now be set out in the next section, will need, where possible, to be sensitive to all these issues.

Part Three

Having set out the valuation framework that will be applied in this study (Part One), and categorised and assessed the available valuation material that corresponds to this framework (Part Two), Part Three will now set out the precise methods that will be deployed in the forthcoming chapters to value virtual water flows. In so doing, Part Three will be directly addressing RQ1.

Section 3.9 begins by looking at the methods that will be applied when estimating the value of off-stream extractive water uses, taking agricultural, industrial and municipal uses in turn. Section 3.10 will address the approach to in-stream values in this study. However, unlike section 3.9, the approach to including in-stream values will be described for this category as a whole rather than addressing the sub-categories (waste assimilation, wildlife habitat and recreation) directly.

3.9. Off-stream values

The three sub-categories of off-stream values are addressed in turn below, starting with agricultural values which, being the most numerous, offer the greatest potential for methodological precision and flexibility.

Agricultural values

As presented in Part Two, agricultural values are the most numerous value category of those recorded with 210 per period value estimates originating from the USA and 145 from countries outside of the USA. Given this, of all the value categories that form part of the analysis here, agricultural values hold the greatest potential for the use of the more advanced BT techniques, and specifically, the estimation of *predictive* regression models. In order to pursue this, the 210 per period values recorded in the USA were selected to form the basis of a regression model. Coming from a single large and diverse country, utilising these values ensured that the data collected for the independent variables (described below) was available in a consistent format across the various sub-national units. Moreover, despite the fact that the majority of the agricultural value estimates collected referred to the western portion of the USA, this potentially incorporated sufficient variation, particularly in terms of climatic conditions, with the arid south west states including Arizona and the more fertile states of the Pacific North West.

As far as the author is aware, there have not been any regression models estimated for agricultural unit values in the peer reviewed literature. However, Scheierling *et al.* (2006) conducted a regression on estimates of the price elasticity of irrigation water demand, and it was the theoretical and conceptual framework set out in that paper which provided the basis of the analysis here. Grounded in production theory, and more specifically the production function, given that irrigation water is a producers use of water and subject to a derived demand, Scheierling *et al.* (2006) looked to explain variations in price elasticity estimates (the dependent variable) using eight categories of analysis (the independent variables). These are: method of analysis, irrigation water price, time frame of analysis (long or short run), farmers' adjustment options (changes in irrigated acreage, crop mix, irrigation technology and schedule), choice of crops (high or low value), type of data (regional or field level), climate (temperature and precipitation) and publication (year of data and peer reviewed or otherwise).

In utilising this framework in the analysis of agricultural unit values here, a number of alterations were made. Firstly, method of analysis (i.e. valuation method employed) was excluded because, as suggested by Brander *et al.* (2012, p.65), 'such variables are not directly applicable in value transfer exercises, i.e. are not used to predict values for new policy sites.' In addition, irrigation water price, farmers' adjustment options, and type of data were also excluded because the sources which had provided the 210 value estimates did not consistently comment on any of these variables. The absence of this level of detail in the unit value literature is something that will be discussed at length in the concluding chapters of this thesis. The absence of these three categories of independent variables, in and of itself, even at this stage, leaves any findings open to criticisms on the basis of omitted variable bias and as such removed the possibility that the analysis was going to provide a useful predictive model. However, the regression was conducted on the basis of the remaining variables as shown in Table 3.26 below to ascertain if any relationships existed which might be worth further study in the future if more detailed unit valuation studies become available.

Table 3.26 Variables used in regression analysis

Variable	Definition of variable
Time frame of analysis (short run or long run)	Dummy variable = 1 for short run and 0 for long run.
Crop value (high or low value)	Dummy variable = 1 for high value and 0 for low value
Climate	
Temperature	Average monthly temperature during growing season (April to October) in study area in (ºF)
Precipitation	Average monthly precipitation during growing season (April to October) in study area (inches)
Publication	
Year of data	Year of value estimate
Peer reviewed journal	Dummy variable = 1 for peer reviewed journal and 0 for non-peer reviewed source.

Time frame, crop value and publication details were taken directly from the studies which provided the estimates used in the analysis. However, temperature and precipitation data, following Scheierling *et al.* (2006), were sourced from the Southern Regional Climate Centre (No date) as the studies did not report this information themselves. A representative town was selected for each study location or, for larger areas, several representative towns, and data (or data averages for larger areas) for temperature (in degrees Fahrenheit) and precipitation (in inches) were used.[19] This data was recorded using a monthly time step and was averaged over the growing season which best represented the crops that were the subject of the value studies (April to October).

The results from the regression are reported in Table 3.27 below. The estimation procedure, again following Scheierling *et al.* (2006), was weighted least squares using weights which were derived from the reciprocal of the square root of the number of estimates from the respective studies. This procedure was used because the 210

[19] For values applicable to whole states, average precipitation and temperature data across all recording stations within that state, was used. Note: some geographical areas were not specific enough to identify appropriate temperature and precipitation data.

estimates were not equally divided amongst the studies they originated from and weighted least squares ensures that one study does not have a disproportionate impact on the results.[20]

Table 3.27 Regression results

	Linear	Double Log	
	(1)	(2a)	(2b)[a]
Intercept	-2305.42 (-1.62)	-21.36 (-1.089)	-19.75 (-1.2)
Time frame of analysis			
Short run (=1)	63.84[b] (6.39)	0.623[b] (4.55)	0.631[b] (5.04)
Crop value			
High value (=1)	80.23[b] (7.18)	0.624[b] (4.21)	0.618[b] (4.35)
Climate			
Temperature	-0.630 (-1.14)	0.533 (1.15)	0.549 (1.22)
Precipitation	-11.22[c] (-2.46)	-0.218 (-1.39)	-0.216 (-1.39)
Publication			
Year of data	1.23[d] (1.72)	0.12 (1.22)	0.011 (1.42)
Peer reviewed (=1)	-47.54[b] (-3.03)	-0.031 (-0.152)	
R^2	0.782	0.665	0.665
Adjusted R^2	0.769	0.645	0.649
Number of observations	107	107	107

[a] Excluding variables with a t statistics lower than an absolute value of 1. Note that the t statistics are uncorrected (see below). [b] Significance at the 1% level. [c] Significance at the 5% level. [d] Significance at the 10% level.

Given the that values for both the dependent variable (unit value estimates in 2014 USD), and the independent variables precipitation and temperature were not normally distributed (see Appendix 17), model two employed a logarithmic transformation of these variables. In the case of the dependent variable, this involved setting the limited number of negative observations to zero, and then adding a constant value of one. For precipitation, where values were all positive but where some were less than one, a constant value of one was added. Only 107 observations were available in the regression model given that the data set evidenced missing data for all independent variables. A large proportion of this missing data was accounted for by the presence of aggregate values for irrigation water which were not crop specific, as well geographic units which were not specific enough to identify appropriate temperature and precipitation data (e.g. Upper Colorado River Basin).

As shown in Table 3.27, whilst the models, as a whole, appear to exhibit reasonable explanatory power (adjusted r square between 67% and 78%), in model 2 only *crop value* and *time frame* are significant ($P < 0.01$). In model 1, *peer review* ($P < 0.01$) and *precipitation* ($P < 0.05$) are also significant. However, given that three of these are

[20] Some studies provided just one estimate. However, several studies provided in excess of 10 estimates, and the largest number of estimates from a single study was 26.

dummy variables and that only one of the scale variables was significant in model 1, the models offer limited potential to predict values in multiple geographies based on variations in variables which could be adjusted for local circumstances, and as mentioned, it is subject to omitted variable bias given that the full range of theoretically derived variables was not available to the analysis. As a result, further exploration of the results was not pursued, including the use of the Newey West procedure[21] that was deployed by Scheierling *et al.* (2006), because, as here, multiple studies that were part of the analysis provided more than one estimate of the dependent variable. Nonetheless, these results may prove useful for future research should additional unit values of agricultural water become available which take into account the full range of theoretically derived variables set out by Scheierling *et al.* (2006).

Separate to the above regression modelling, water stress was also examined as a potential explanatory variable in its own right given its use in the work of Trucost (see Chapter Two), albeit in the context of in-stream values, and Park *et al.* 2015. However, as mentioned in Chapter Two, whilst not devoid of theoretical foundation, it should be noted that this approach is not grounded in an *encompassing* theoretical framework such as the production function utilised above.

Baseline water stress data, on a basin scale, was sourced from the World Resources Institute (2013) for each of the geographies that the 210 agricultural value estimates corresponded to. More specifically, where the value estimate was site specific, the water stress value for the basin within which the site was located was utilised. Where the value estimate was not site specific but instead referred to a state within the USA, or where the estimate referred to a water region which was comprised of multiple basins, water stress values were converted using Geographical Information Systems ArcGIS v.10.4.1. This involved calculating the average water stress value for these larger geographic units based on the basins that fell within their boundaries, using river basin area as the weighting factor. Table 3.28 below provides an overview of the results from the regression analysis. As shown, it is clear that in the context of the agricultural values assessed, baseline water stress does not appear to be a predictor of agricultural values.

[21] This procedure attempts to correct ordinary least squares standard errors in the presence of autocorrelation and heteroscedasticity of unknown form.

Table 3.28. Regression modelling results using baseline water stress as the single predictor variable

Model	Adjusted R^2	Significant at 1%	Significant at 5%	Significant at 10%
Linear	0.003	No	No	No
Semi log [a]	0.006	No	No	No
Double log	0.02	No	Yes	
Linear quadratic [b]	0.041	Yes		
Log quadratic [b]	0.057	Yes		

[a] Untransformed dependent variable. [b] Quadratic applied to independent variable.

Given the results from the regression modelling, it is quite clear that there is too much variation in the agricultural values category to make anything other than single point BT, and the careful selection of individual estimates, viable. This conclusion accords with Gibbons (1987, p.39) who suggested, in the context of their review of unit value estimates 30 years ago, that 'geographic variation appears to be lost in the noise of a host of different statistical methodologies and assumptions about technology, crop mix, and time frame.' The principal implication of this is that only those geographies where a unit value estimate already exists – see Tables 3.10 and 3.12 – or neighbouring geographies with similar characteristics, can be covered by the method developed here. This is obviously an important limitation, however, by utilising single point BT within the same country (or transferring to similar neighbouring countries), the advantage over any regression model is that the values generated will not need *post hoc* adjustment. For example, if the regression model described previously had yielded robust estimates of irrigation water, in order to transfer these to geographies outside the USA, they would need to have been adjusted, perhaps to reflect relative agricultural prices in the country that they were being transferred to. However, appropriate adjustments to agricultural values have not been covered in the literature to date. This brings us to the protocol which will be deployed for transferring individual estimates which will be covered below.

Single point BT protocol (agricultural values)

There are numerous protocols which have been developed for guiding the use of benefits transfer (for example, see Navrud, 2007). However, what they all have in common is that they have been developed for use when transferring values in situations where there are overt dissimilarities between the study and policy sites, particularly in terms of the environmental good in question and the nature of the sites themselves. Indeed, most protocols have been developed with particular reference to recreational values which have been estimated using the travel cost and contingent valuation methods. These protocols, amongst other things, consider divergences in terms of the affected populations and their socio-economic characteristics, the physical characteristics of the goods involved, the scale of the change being valued, and the presence or absence of substitutes. However, in the context of agricultural values, which as we have seen are derived from private goods and which have been predominantly estimated using relatively simple valuation techniques, such considerations are not directly applicable. Therefore, whilst the protocol developed here takes into account existing best practice, it focuses only on those aspects which are relevant in this context. Some of these have already been covered in Part Two in the discussion of how values were selected and updated. However, three further considerations, as suggested by Johnston and Rosenberger (2010), are detailed below:

1. Considerations of primary study measurement error,
2. Considerations of generalisation or transfer errors and,
3. Defining a consistent scenario.

On the first of these, Part Two set out at length the criteria that were used in order to select the sources for summary and analysis, including agricultural values. These ensured that all sources catalogued utilised appropriate methods and generated appropriate and identifiable values, and thus, the scientific soundness of the pool of agricultural values from which transfers can occur. In addition, in each of the case studies chapters that follow, the quality of the analysis used to generate the values used for transfer will also be directly addressed.

In terms of the second area and generalisation error, Rosenberger and Loomis (2000, p.1097) identify two convergent validity tests that are applicable to BT. First, comparing the transferred value to a ‘true’ value at the policy site which has been estimated using primary valuation techniques, and second, comparing two different transfer estimates to ensure that judgments by the analyst do not influence the conclusion. To this could also be added that estimates from SP methods can also be compared to estimates from RP methods and vice versa. However, given the limited number and uneven spread of the agricultural value estimates catalogued, tests of convergent validity are not feasible in this context, and indeed, have limited application in single point BT. Indeed, outside the USA, there are insufficient values for any country to be able to compare the transferred value with a true value, or, to compare the transferred value with one estimated using a different class of technique. As such, the method employed here will make use of a common technique in economic analysis – sensitivity analysis – to understand how sensitive any conclusions reached are to changes in unit values. In addition, it should be noted that as Johnston and Rosenberger (2010, p.486) argue, precision in BT is a function of the significance of the policy decision, with '… higher degrees of precision and consequently lower transfer errors needed…as one moves from broad benefit-cost analyses for information gathering or screening of projects and policies to calculation of compensatory amounts in negotiated settlements and litigation.' Therefore, given that the method here is, as discussed, looking simply to provide high-level insight, then a higher level of transfer error becomes acceptable.

Finally, the third area of the protocol involves defining a consistent scenario 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. This will be commented on in each of the following case studies, but means ensuring that the water value type (at source/at site, long run/short run, high valued crops/low valued crops) 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.

In terms of the green water that is consumed during the agricultural stages of the supply chains, it had been anticipated that values for artificially applied irrigation water that was *consumed* by the crop would be used as a proxy. However, it is quite clear following Part Two that there is a dearth of this value type. Therefore, the case studies will make use of a lower bound estimate of water consumption where necessary, such as water application, which has been measured net of extraction costs (i.e. an at source value) which would seem to be most appropriate for green water. However, the specifics of the values used for green water will be commented on in each case study.

Industrial values

In Part Two it was argued that when it comes to methodological development in the area of industrial water values, there are four studies in particular, all conducted outside of the USA, that represent the most advanced sources in what is still a relatively unstudied field (Renzetti and Dupont, 2002; Wang and Lall, 2002; Kumar, 2004; Bruneau, 2007). In this context, the unit values estimated by Wang and Lall (2002) and Bruneau (2007) will be drawn on to estimate the value of water used directly in the food industry in Chapters Four and Five (the operational water footprint). These two sources have been chosen because they meet the following criteria:

1. They both estimate unit values for the food industry specifically,
2. The estimates are for water consumed which accords with the approach to water measurement used in water footprint accounting as applied here, and
3. They have both been conducted in, and are relevant to, advanced economies.

The study by Bruneau (2007) utilises an alternative cost approach to estimate the *average* shadow value of water in industry in Canada. In this, the cost of water recirculation or recycling is a substitute for additional intake water. However, as Renzetti and Dupont (2002:4) argue, recycled water may be of lower quality than raw intake water, or, it may produce benefits for the firm such as reclaimed heat or avoided effluent charges i.e. raw intake water and recycled water may not be perfect substitutes. Therefore, when deploying the values from Bruneau (2007) in Chapters Four and Five, it is necessary to recognise that there are potential limitations in the method applied, but also that Bruneau (2007) is one of only two sources (the other being Wang and Lall, 2002) that meets all the criteria mentioned above. Wang and Lall (2002) utilise a production function to calculate the *marginal* value of water in industry in China. The critique of their work, as set out in Part Two, is also provided by Renzetti and Dupont (2002). However, as stated previously, Wang and Lall (2002) appear to be the only source which has attempted to provide an estimate of the physical productivity of a unit of water in industry and, along with Renzetti and Dupont (2002) and Kumar (2004), represent the most advanced approach to estimating industrial water values that is currently available in the literature.

Table 3.29 below presents the value of water consumed in the food industry as estimated by Wang and Lall (2002) and Bruneau (2007). As shown, whilst the methods used in the two studies differ, and yield different value conceptions, the value estimates are relatively similar.

Table 3.29 Food industry values

Source	Method	Value type	Water volume measure	Original value m^3 (currency)	2014 USD per 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

Separate to the direct use of water in the food industry (the operational water footprint), there is also the water accounted for by industry which falls under the supply chain overhead water footprint and the water footprint associated with various packaging inputs. As presented in Part Two, given that industrial values were found to be specific to the industrial water use (with the associated water quality requirements) it is not feasible to place a value on broad categories such as the supply chain overhead water footprint which encompasses numerous items i.e. they are not functionally specific. However, as argued in Part One, given that many of these items are sourced from world markets, they will never be a relevant change variable when comparing water values in different regions.

Municipal values

As presented in Part Two, municipal values are both relatively few in number and unevenly distributed geographically. Indeed, as we will see in the following chapters, there are no applicable value estimates for the two locations – one in Chapter Four and one in Chapter Five – which refer to the water used by end consumers when either drinking tea or cooking pasta. Moreover, it was noted previously that a large proportion of the municipal values in the USA have been derived by using a simple, and easily replicable, formula for a household demand function which draws on a few pieces of easily obtainable information, namely an initial price level, an indication of volumetric usage, and an estimate of the price elasticity of demand. Given these factors, the approach to placing a value on municipal water use adopted here will make use of this simple household demand function as set out in Part Two section 3.7.3. This function provides a recognised and bespoke means of valuing household water use which is preferable to any attempt to transfer values from the fragmented pool of municipal value estimates set out in Part Two. Specifically, the household demand function requires a specified reduction in volumetric usage in order to provide a second point on the demand curve and thus estimate a unit value. In this context, it will be assumed, in all instances, that this will be a 10% reduction in volume which is in line with the approach adopted by Gibbons (1987).

3.10. In-stream values

The in-stream values catalogued in Part Two exhibited a number of limitations in this context. In particular, it is clear that there are only a handful of waste assimilation and wildlife habitat values, and in the case of the latter, they originate predominantly from the south west corner of the USA. Consequently, there does not appear to be a means of reliably estimating waste assimilation and wildlife habitat values across differing geographies. With regard to recreational values, given the number of estimates collated (second most studied area behind agriculture), in principle it seems that meta-analytic BT could be attempted with the pool of value estimates gathered in order to *predict* values in geographies other than the USA.

However, as mentioned in Part Two, since the unit value of water for recreational purposes, correctly conceived, is driven primarily by varying levels of flow, only a sub-set of the recreational value data would be available to generate a pooled regression model. Nonetheless, provided that the size and profile characteristics of the rivers covered in the respective studies are controlled for given that, for example, low flow levels on one river might represent high flow levels on another, and vice versa, a regression analysis may be viable. Brown (1991) provides the most comprehensive overview to date of the recreational value literature which has explicitly derived recreational values based on *specified* levels of river flow in Cubic Feet per Second (CFS). The studies mentioned in Brown (1991), together with those additional studies found during the literature search in Part Two, which are all applicable to the USA, are summarised in Table 3.30 below.

The problems with conducting a regression analysis on the literature in Table 3.30 if the aim is then to predict values in disparate geographies that would be useful in this context, however, are threefold:

1. The level of flow at each supply chain stage will be unknown at the level of spatiotemporal detail that is necessary in this context,
2. Even if the level of flow was available, the question would be whether this should this be measured at the site of the supply chain stage, or the broader basin within this sits, and
3. Linked to the above, and most significantly, the *distance decay effect* will also be unknown.

Table 3.30 Recreational value studies based on variations in river flow

Author (paper reference number)	USA State (River)	Flow variation (CFS)	Recreation activity	Valuation approach
Bishop *et al.* (4)	AZ (Colorado River)	Low flow 10,000	Rafting	CVM
Daubert and Young (22)	CO (Poudre River)	100 - 700	Fishing	CVM
Daubert and Young (23)	CO (Poudre River)	50 - 700	Fishing	CVM
Duffield *et al.* (24)	MT (Big Hole and Bitterroot rivers)	100 - 2000	Predominantly fishing	CVM (DCE)
Gibbons (29)	WA (Yakima River System)	Minimum flows 805	Fishing	Unspecified
Harpman (33)	CO (Taylor River)	Critical winter low flow 40	Fishing	CVM
Johnson and Adams (40)	OR (John Day River)	Mean summer flow 204 and mean spring flow 2,700	Fishing	CVM
Loomis and McTernan (49)	CO (Poudre River)	500 - 2,300	Non-commercial kayakers and river rafters	CVM and TCM
Narayanan (57)	UT (Blacksmith Fork River)	Low flow 80	Camping, hiking and fishing	TCM

The distance decay effect, in simple terms, means that people are more likely to be WTP for recreation the closer they are to the site in question (Pate and Loomis, 1997; Hanley *et al.* 2003). It is a feature which is peculiar to recreational values due to the different methods that are used to estimate waste assimilation values (alternative cost) and wildlife habitat (water market transactions) which provide a value of that water *in situ*. In this context, unknown distance decay effect makes it prohibitively difficult to reliably estimate recreational values across geographies. As a result, Chapters Four to Six will not look to estimate recreational values, or any other in-stream values, directly, but rather, will focus on off-stream values. However, theses chapters will include a number of sensitivities in order to understand how sensitive any conclusions reached are to changes in unit values, and as part of this, the possibility that in-stream values might account for these changes will be commented upon. In order to do this, Table 3.31 below presents an in-stream value scale which is based on the in-stream value estimates collected and set out in Part Two. This shows the minimum, median and maximum in-stream values that were collected <u>assuming that waste assimilation, wildlife habitat and recreation were all present at the same time and in the same location, that the point of diversion is such that the values are all additional (see Figure 3.3 above), and that there is no distance decay effect for recreational values.</u> For example, the maximum in-stream value on the scale is based on the highest recorded unit values for waste assimilation, wildlife habitat and recreation.

The utility of the scale comes from the fact that, as we will see, many of the values that the sensitivities will derive go far beyond the maximum value on the scale which, based on the evidence collected, is the ‘worst case’ or most extreme scenario. Indeed, the values for wildlife habitat and recreation, in particular, have predominantly been estimated in the arid parts of the south western USA, which is the region that has, by necessity, most explored the unit valuation of water as a means of improving inter-sectoral water allocation. Therefore, in these circumstances, it seems safe to conclude that the presence of in-stream values would be unlikely to alter the conclusions reached.

Table 3.31 In-stream value scale (USD 2014 per m3)

Low		Median		High
$ 0.0006	→	$ 0.06	→	$ 0.6

When applied at each supply chain stage, the values on the scale will need to be adjusted for relative incomes in order for them to be relevant in each geography. This will be done by using the formula set out by Czajkowski and Scasny (2010) which assumes an income elasticity of one:

$$WTP_{ps} = WTP_{ss}\left(\frac{\overline{I_{ps}}}{\overline{I_{ss}}}\right)\varepsilon$$

where WTP_{ss} is willingness to pay at the study site, WTP_{ps} is the willingness to pay estimate transferred to the policy site, and I_{ss} and I_{ps} are mean income levels at the study and policy sites. ϵ represents the income elasticity of willingness to pay between the mean income levels at the study and policy sites which is assumed to be one (Czajkowski and Scasny, 2010).[22]

3.12 Summary (Part Three)

In summary, Chapter Three began in Part One by setting out those aspects of the methodology that were not contingent upon the precise method that will be used to value virtual water flows. This included the ESS valuation framework that guided the detailed

[22] The study by Czajkowski and Scasny (2010) suggests that using an income elasticity of WTP of one, which they note is the ‘usual choice for income adjustments with no other information,’ is most appropriate when the countries are highly heterogeneous in income levels.

review of the unit value literature in Part Two. Based on the analysis in Part Two, Part Three has set out a method for the valuation of virtual water flows, thus directly addressing RQ1 and the extent to which the valuation literature can support this aim. As described above, the method is constrained by the availability and nature of the values that were catalogued, and thus primarily focuses on the direct use, or off-stream, value of water along the supply chain. The following chapters now look to the application of this method, firstly in the context of the durum wheat pasta (Chapter Four) and tea (Chapter Five) supply chains, both of which utilise secondary data in order account for the volumes of virtual water, as well as the potato crisp supply chain (Chapter Six) which is based on an original water footprint study conducted by the author.

4. The durum wheat pasta supply chain

This chapter sets out the durum wheat pasta supply chain case study, volumetric water data and supporting information for which has been obtained from secondary sources as detailed throughout.

Part A summarises the durum wheat pasta water footprint i.e. the volumes of green and blue consumptive water use, and degradative grey water use, at each point along the supply chain. Part B summarises the attendant monetary values that have been assigned to these volumes of water based on the approach set out in Chapter Three.

Part A – The pasta water footprint

Part A begins by setting out the production unit that is the subject of analysis in this chapter, and providing an overview of the associated supply chain map (section 4.1). Section 4.2 sets out the consumptive blue and green water use, and degradative grey water burden, for the *supply chain* water footprint directly associated with inputs. Section 4.3 repeats this for the *operational* water footprint directly associated with inputs. Section 4.4 details the assumptions made regarding the consumptive blue water use during the consumer use phase (i.e. the water used by the end consumer during cooking). Section 4.5 details those aspects of the analysis which are out of scope. Finally, section 4.6 summarises the total water footprint of durum wheat pasta.

4.1. Product units and supply chain map

This case study is loosely based on the analysis by Ruini *et al.* (2013) who examined the water footprint associated with the production of 1 kilogram of durum wheat pasta by the food company Barilla ("the company"), who claim to be the world's largest user of durum wheat in the world (Barilla, 2015). However, as detailed in section 4.2, several realistic adaptations have been made in this context – principally around the precise locations where durum wheat is sourced from – in order to introduce additional geographical variation, that was previously unaccounted for in the analysis by Ruini *et al*. (2013), for analysis and testing of the valuation methodology in Part B.

Ruini *et al.* (2013) describe the pasta supply chain as encompassing three principal stages, each of which impacts on fresh water resources: the cultivation of durum wheat (stage 1), factory based milling and pasta processing (stage 2), and finally the consumption of pasta by the end consumer (stage 3). Whilst the company sources wheat from, and mills and processes pasta in, multiple locations globally, the supply chain that is isolated here from those presented in Ruini *et al.* (2013) is focused on the company's Italian production facilities (stage 2). These have been chosen over those facilities in the USA, Turkey and Greece because Italy is the leading global exporter of pasta (accounting for 32% of the global market in 2013) and as such, sources durum wheat from a broad range of countries, a level of variation that will provide a rich background for the analysis in Part B (Simoes and Hidalgo, 2011). Indeed, the origin of the durum wheat used in the company's Italian production facilities includes Canada (8%), the USA (5%), Mexico (6%), Italy (70%), France (10%) and Greece (1%) (Ruini *et al.*, 2013). The consumer use phase at stage 3 is assumed to occur in Germany which accounts for the second largest share (21.7%) of *exports* from the company's Italian production facilities (the majority or 63.2% is consumed domestically). Germany has been chosen for analysis over France (28%), because it introduces a dissimilar but realistic point of geographical variation for analysis in Part B (Ruini *et al.*, 2013).

Figure 4.1 below sets out the supply chain map for durum wheat pasta.

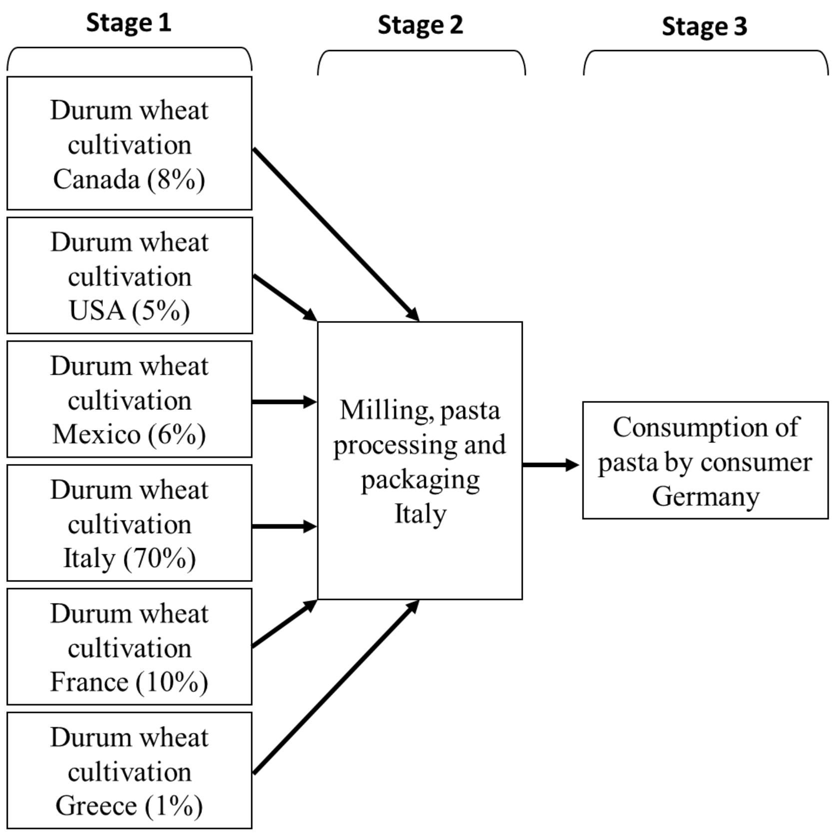

Figure 4.1. Durum wheat pasta supply chain map (based on Ruini *et al.* 2013). Note: Percentage figures for durum wheat cultivation during stage 1 refer to the origin of durum wheat used in the company's Italian production plant in stage 2.

4.2. Supply chain water footprint directly associated with inputs

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.[23] Table 4.1 below, using data from the *Water Stat* database, sets out the water footprint of durum wheat in 18 separate locations which span the six countries of origin mentioned in section 4.1 (Mekonnen and Hoekstra, 2010a).

Table 4.1. Water footprint of durum wheat for selected country and region combinations (m^3 per tonne)

Country	State/region	Green water	Blue water	Grey water	**Total**
Canada	Alberta	1,247	16	202	**1,465**
Canada	Saskatchewan	1,369	1	206	**1,576**
USA	Arizona	399	848	156	**1,403**
USA	California	726	522	158	**1,406**
USA	Montana	2,354	53	299	**2,706**
USA	North Dakota	1,256	1	184	**1,441**
Mexico	Baja California	341	325	186	**852**
Mexico	Sonora	249	432	184	**865**
Italy	Basilicata	1,342	15	202	**1,559**
Italy	Calabria	1,440	17	213	**1,670**
Italy	Campania	1,271	11	189	**1,471**
Italy	Puglia	1,372	42	212	**1,626**
France	Centre (Orleans)	587	2	6	**595**
France	Midi-Pyrenees (Toulouse)	708	4	6	**718**
France	Languedoc-Roussillon (Montpellier)	798	4	7	**809**
Greece	Western Macedonia (Kozani)	1,356	2	119	**1,477**
Greece	Central Macedonia (Thessaloniki)	1,477	35	141	**1,653**
Greece	Thessaly (Larissa)	1,442	34	139	**1,615**

Source: Mekonnen and Hoekstra (2010a).

The data in Table 4.1 has been used here, in preference to that reported by Ruini *et al.* (2013), because the latter report water use during crop cultivation as a weighted average, for each country where their stage 2 production facilities are located, based on the origin of the wheat that is used i.e. the water use during crop cultivation is not split out by geographical location, as it needs to be for the analysis in Part B. Moreover, the durum wheat water footprint data used by Ruini *et al.* (2013) to create these weighted averages

[23] Aldaya and Hoekstra (2010) suggest that salt is also present in the production of durum wheat pasta. However, they exclude it from their analysis on the basis that it has an immaterial impact on the water footprint. In the case of Ruini *et al.* (2013), salt is either not a component of their durum wheat pasta recipe, or it has likewise been excluded on the basis of materiality. Given this, the possible presence of salt has also been excluded in this context.

appears to be based on country level, rather than regional specific data, and as such, it does not take account of variations in the principal durum wheat growing areas within the six countries at stage 1, which we are able to do here. Indeed, the rationale behind the selection of the 18 locations analysed here is that in the case of Canada (Canadian Grain Commission, 2016), the USA (USDA, 2016), Mexico (USDA Foreign Agricultural Service, 2016), Italy (USDA Foreign Agricultural Service, 2012), France (France AgriMer, 2011) and Greece (USDA Foreign Agricultural Service, 2010), they represent the principal durum wheat growing regions in these respective countries. In the case of the USA and Canada, detailed regional statistics are available regarding durum wheat production and are shown in Figures 4.2 and 4.3 below.

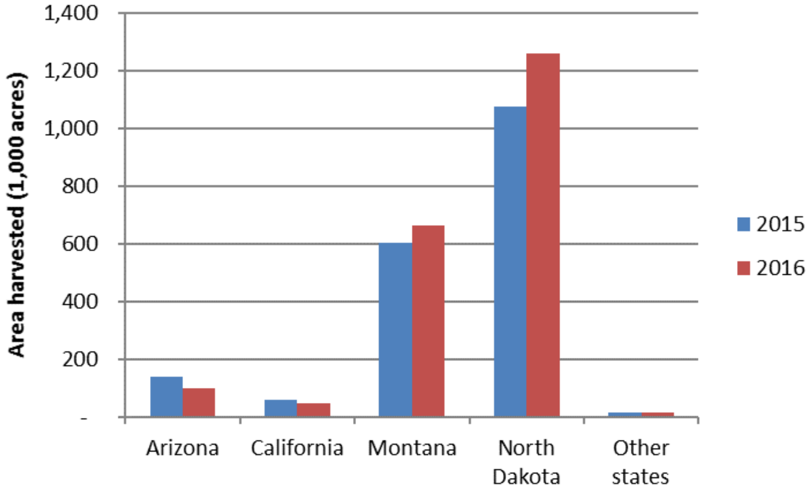

Figure 4.2. Durum wheat area harvested in the USA by state (source: USDA, 2016).

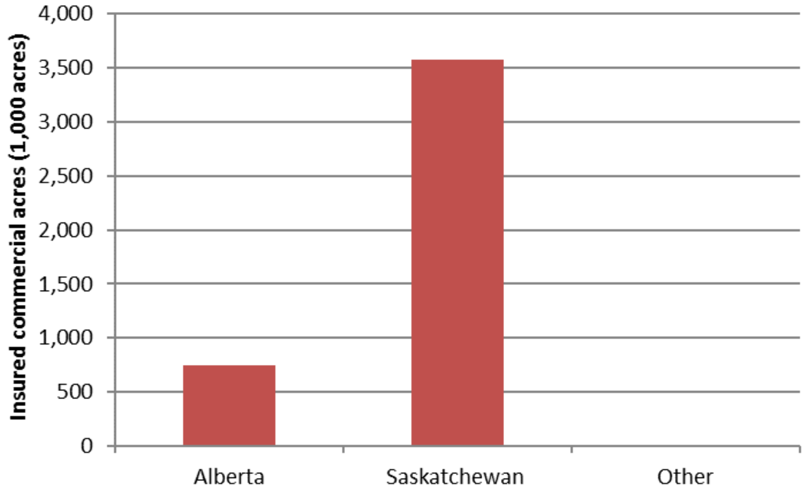

Figure 4.3. Canadian amber durum 2016 insured commercial areas (source: Canadian Grain Commission, 2016).

As shown in Table 4.1, the total water footprint of durum wheat ranges from 2,706 m^3 per tonne in Montana (USA), to 595 m^3 per tonne in Orleans (France), with an average across the 18 locations of 1,384 m^3 per tonne. In the northern USA states of Montana and North Dakota, Canada, France and, to a lesser extent, Greece and Italy, durum wheat is a predominantly rain fed crop, whereas in California, Arizona and the two Mexican states, significant irrigation occurs. Indeed, in Arizona and the two Mexican states, blue water represents a larger share of the total water footprint than green water. In terms of the differences between regions, given that the blue and green water footprint figures per tonne are derived from total evapotranspiration per hectare divided by the crop yield per hectare (ET/Y) (see equations five to eight in Chapter Three), disparities in the figures can be explained by the interplay of both these variables. However, it is variations in crop yield which explains the most obvious differences in Table 4.1. For example, the water footprint in Mexico is noticeably smaller than in the USA and Canada because the average wheat yield in the former (5.2 tonnes per hectare) is significantly large than the latter (2.9 tonnes per hectare) (FAOSTAT, 2016), itself stemming from the fact that wheat is irrigated in Mexico which has the effect of boosting yields (Mekonnen and Hoekstra, 2010c). Similarly, the comparatively low water footprint in France stems in large part from the fact that the national average yield is 7.4 tonnes per hectare.[24]

Regarding grey water, it should be noted here that the figures in the *Water Stat* database are based on a Nitrogen fertiliser only and assume a leaching rate of 10%, natural nitrogen concentrations of zero, and a nitrogen water quality standard of 10 mg/l (Mekonnen and Hoekstra, 2010a).

In line with the approach adopted by Aldaya and Hoekstra (2010), it is assumed here 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 (see Figure 4.4 below). Drawing on these assumptions, the water footprint of semolina can be derived. Using the average water footprint of durum wheat across the 12 locations in Table 4.1 as an example (i.e. 1,384 m^3 per tonne), the average water footprint of semolina is 1,692 m^3 per tonne ((1,384/0.72) $\times$ 0.88).

[24] Note that data in the *Water Stat* database was estimated based on national average yield data (Mekonnen and Hoekstra, 2010a).

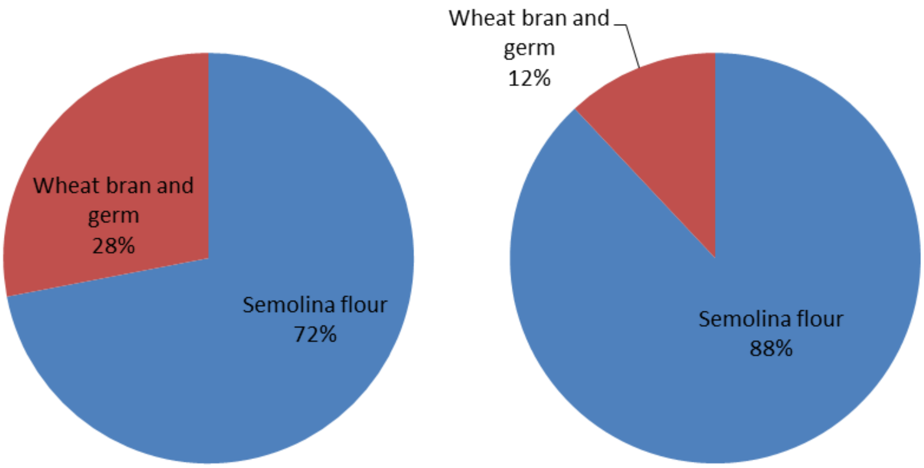

Figure 4.4. Durum wheat product fraction and value fraction (from left to right) (Aldaya and Hoekstra (2010).

Table 4.2 and Figure 4.5 below set out the water footprint of semolina flour, processed from durum wheat sourced from each of the 18 locations analysed, based on the product and value fractions in Figure 4.4.

Table 4.2. Water footprint of semolina for selected country and region combinations using a product fraction of 72% and a value fraction of 88% (m^3 per tonne)

Country	State/region	Green water	Blue water	Grey water	**Total**
Canada	Alberta	1,524	20	247	**1,791**
Canada	Saskatchewan	1,673	1	252	**1,926**
USA	Arizona	488	1,036	191	**1,715**
USA	California	887	638	193	**1,718**
USA	Montana	2,877	65	365	**3,307**
USA	North Dakota	1,535	1	225	**1,761**
Mexico	Baja California	417	397	227	**1,041**
Mexico	Sonora	304	528	225	**1,057**
Italy	Basilicata	1,640	18	247	**1,905**
Italy	Calabria	1,760	21	260	**2,041**
Italy	Campania	1,553	13	231	**1,797**
Italy	Puglia	1,677	51	259	**1,987**
France	Centre (Orleans)	717	3	7	**727**
France	Midi-Pyrenees (Toulouse)	866	5	7	**878**
France	Languedoc-Roussillon (Montpellier)	976	5	8	**989**
Greece	Western Macedonia (Kozani)	1,657	3	145	**1,805**
Greece	Central Macedonia (Thessaloniki)	1,805	42	172	**2,019**
Greece	Thessaly (Larissa)	1,762	41	169	**1,972**

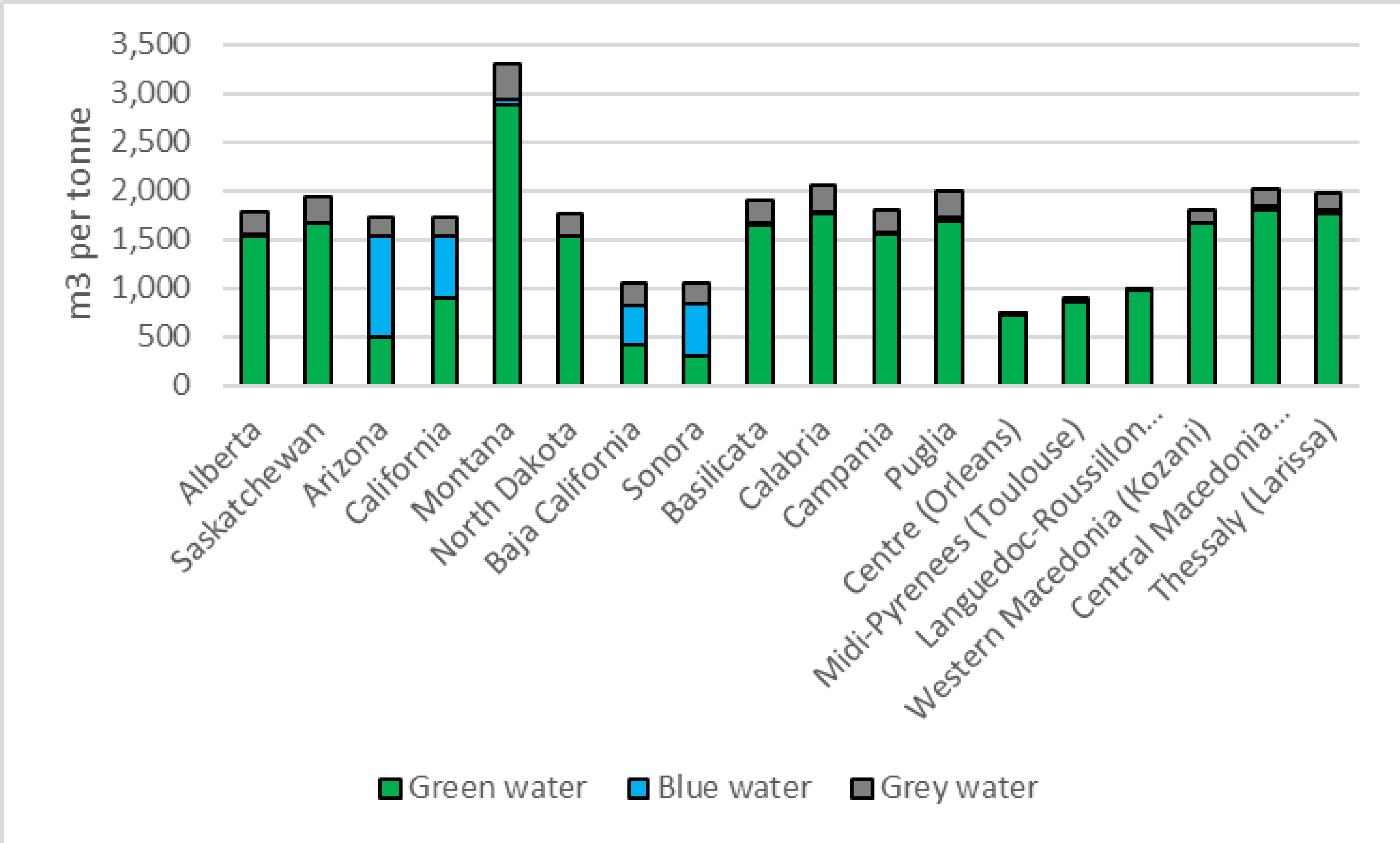

Figure 4.5. Water footprint of semolina for selected regions (Mekonnen and Hoekstra, 2010a).

In addition to the water footprint associated with durum wheat/semolina, the supply chain water footprint directed associated with inputs also includes the water burden linked to the primary and tertiary packaging. Ruini *et al.* (2013) estimate that this accounts for two litres of blue water per kilogram of pasta. As noted in Chapter Three, given that the water burden associated with packaging inputs is geographically non-specific, it is assumed here that these two litres of blue water are extracted in the same location as the factory at stage 2 which makes use of these packaging inputs (see below).

4.3. Operational water footprint directly associated with inputs

The operational water footprint directly associated with inputs includes the water used at stage 2 in milling and pasta processing, all of which is assumed to occur in Pedrignano in Northern Italy. Of the company's five production locations, this is the only one which combines a production plant and a mill in a single site.

The steps in pasta production include pre-cleaning and tidying up, conditioning, milling, raw material storage, mixing dough and rolling, drying, packaging, storage and distribution (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. Based on data from the company's five Italian production sites (two mills, two processing plants and one joint mill and plant), Ruini *et al.* (2013) suggest that the average water footprint of these main production steps is approximately four litres per kilogram of pasta, all of which is blue water.[25] This figure includes both the water used directly in milling and processing, as well as that linked to the associated energy and transportation requirements. Any wastewater produced during pasta production is assumed to be returned, via a public sewage system, to a waste water treatment plant. Therefore, the grey water footprint is assumed to be zero.

4.4. The water footprint of pasta consumption

In line with Ruini *et al.* (2013), it has been estimated that it takes approximately ten litres of water to cook a kilogram of pasta. Whilst not all of these ten litres will register as consumptive blue water use given that only a portion of the water will evaporate during cooking, owing to the difficulty of approximating evaporative use only, the full amount

[25] This is a blended average based on the contribution that each mill and plant makes to total production.

has been assumed. Any wastewater associated with pasta consumption is again assumed to return to a waste water treatment plant and thus there is no grey water footprint associated with pasta consumption. As mentioned, it is further assumed that final consumption of the pasta take place in Germany (Berlin).

4.5. Out of scope and caveats

Both the operational and supply chain overhead water footprints have been excluded from the analysis here due to a lack of specific data in Ruini *et al.* (2013) and because, particularly for agriculture based supply chains, they tend to be immaterial when compared to the volumes of water used to produce the product (see tea and potato case studies in this thesis as well as Ercin *et al.*, 2011 and Jefferies *et al.*, 2012). The water footprint of labour has also been excluded in line with the established methodology set out in the Water Footprint Assessment Manual (Hoekstra *et al.*, 2011). This is to prevent double counting given that workers are also consumers.

It should be noted here that it is unclear whether Ruini *et al.* (2013) utilised a similar approach to the product and value fractions mentioned in section 4.2 and accounted for the multiple output products that are derived from durum wheat (i.e. semolina and wheat bran and germ). Doing so, in this case, has the effect of increasing the water footprint for semolina, as can be seen by comparing Tables 4.1 and 4.2, when compared to durum wheat alone. The significance of this is that is that if they have not made a similar determination, then the four litres of water used during stage 2 would need to be multiplied by the value fraction (88%) in order to apportion this water volume between semolina and wheat bran and germ.[26] However, given that this makes an immaterial difference in this context (3.52 litres versus 4 litres) the value fraction has not been applied to the water used during stage 2.

4.6. Total water footprint

The total water footprint for one kilogram of durum wheat pasta is shown, for two scenarios, in Tables 4.3 and 4.4 below. Each scenario is based on the blend of durum wheat sources that was noted in Figure 4.1 and section 4.1. However, for each of the six countries that durum wheat is sourced from at stage 1, it is assumed in scenario one that

[26] Given that the four litres of water is per kilogram of *processed* product, it would not need to be divided by the product fraction as well. If it had been recorded per unit of *input* product, it would have needed to have been divided by the product fraction and then multiplied by the value fraction.

the region within that country with the highest water footprint provides the durum wheat crop. Conversely, for scenario two, it is assumed that the region with the lowest water footprint provides the durum wheat crop. Where one of the countries at stage 1 only had two regions in the 18 region set, then both were used (i.e. Canada and Mexico). However, where the country had more than two regions (USA, Italy, France and Greece) then utilising the highest and lowest *total* water footprints in scenarios one and two managed to also capture the highest and lowest green, blue and grey water use in crop cultivation. The one exception to this is durum wheat grown in Italy where the highest region (Calabria) and lowest region (Campania) did not encompass the region with the highest level of blue water use in crop cultivation (Puglia). Nonetheless, in addition to valuing the water used in scenarios one and two, Part B will also look at the value of water in each of the 18 regions where wheat is grown.

Note that data for stages 2 and 3 remains the same in scenario one and two. In addition, the figures in Table 4.3 and 4.4 below can also be read as cubic metres per tonne which will aid the valuation exercise in Part B given that values for water tend only to register in higher volumetric measures. Indeed, for this reason, in this case study and those that follow, one tonne of finished goods (excluding packaging) will be the primary unit that is used for valuation purposes.

Table 4.3. Scenario one (high) – maximum water footprint 1kg of durum wheat pasta (litres)

Supply chain stage	Location	Description	Water footprint component	Green	Blue	Grey	**Total**	% of total
1 [a]	Canada (Saskatchewan)	Durum wheat/semolina	Supply chain	134	0	20	**154**	8
1 [a]	USA (Montana)	Durum wheat/semolina	Supply chain	144	3	18	**165**	8
1 [a]	Mexico (Sonora)	Durum wheat/semolina	Supply chain	18	32	13	**63**	3
1 [a]	Italy (Calabria)	Durum wheat/semolina	Supply chain	1,232	15	182	**1,429**	73
1 [a]	France (Languedoc-Roussillon)	Durum wheat/semolina	Supply chain	98	1	1	**100**	5
1 [a]	Greece (Central Macedonia)	Durum wheat/semolina	Supply chain	18	0	2	**20**	1
2 [b]	Italy (Pedrignano)	Packaging	Supply chain	0	2	0	**2**	>1
2 [b]	Italy (Pedrignano)	Milling and pasta processing	Operational	0	4	0	**4**	>1
3 [b]	Germany	Pasta consumption	Pasta consumption	0	10	0	**10**	>1
Total				**1,644**	**67**	**236**	**1,947**	**100**

Source: [a] Mekonnen and Hoekstra (2010a). [b] Ruini *et al.* (2013).

Table 4.4. Scenario two (low) – minimum water footprint 1kg of durum wheat pasta (litres)

Supply chain stage	Location	Description	Water footprint component	Green	Blue	Grey	**Total**	% of total
1 [a]	Canada (Alberta)	Durum wheat/semolina	Supply chain	122	2	20	**144**	9
1 [a]	USA (Arizona)	Durum wheat/semolina	Supply chain	24	52	10	**86**	5
1 [a]	Mexico (Baja California)	Durum wheat/semolina	Supply chain	25	24	14	**63**	4
1 [a]	Italy (Campania)	Durum wheat/semolina	Supply chain	1,087	9	162	**1,258**	76
1 [a]	France (Centre)	Durum wheat/semolina	Supply chain	72	0	1	**73**	4
1 [a]	Greece (Western Macedonia)	Durum wheat/semolina	Supply chain	17	0	1	**18**	1
2 [b]	Italy (Pedrignano)	Packaging	Supply chain	0	2	0	**2**	>1
2 [b]	Italy (Pedrignano)	Milling and pasta processing	Operational	0	4	0	**4**	>1
3 [b]	Germany	Pasta consumption	Pasta consumption	0	10	0	**10**	>1
Total				**1,347**	**103**	**208**	**1,658**	**100**

Source: [a] Mekonnen and Hoekstra (2010a). [b] Ruini *et al.* (2013).

As can be seen from Tables 4.3 and 4.4, sourcing wheat from those regions which consume and degrade the least *total* water volume in each of the six countries provides a saving of nearly 300 litres per kilogram of pasta (or 300 m^3 per tonne). However, scenario two utilises significantly more of globally limited blue water resources. In order to choose between scenarios such as this, and go beyond simply accounting for water volumes, as we have seen, water footprinting takes in to account the vulnerability of water systems using the water stress index which measures the ratio of total annual water withdrawals in an area to total annual water availability. Table 4.5 below sets out the water stress values for each of the sourcing locations at stage 1 (note there was too much variation in water stress values across the two large regions in Canada to identify a relevant value).[27] These can be used to assess the impact of blue water usage in the supply chain and thus identify 'hotspots.' Following the approach set out in Jeffries *et al.* (2012, p.159), a hotspot occurs where 'the blue water footprint of products is large and where water scarcity is high,' the latter being defined as where it exceeds a value 0.6. In this context, this would suggest that each of the four Italian regions are hotspots given the respective water stress values and the fact that Italy provides 70% of all durum

[27] Baseline water stress values in Table 4.5 for the USA are specific to the durum wheat growing regions in each state. For Italy, the water stress values are those which apply in the regional capital. In all other instances, water stress values are representative of the geography specified.

wheat used in the end product. In addition, whilst Mexico and the USA provide only 5-6% of the durum wheat used, water stress values in excess of 1 for Sonora, Baja California, Arizona and California, meaning that more water is withdrawn than is available, suggest that these regions also represent potential hotspots. However, choices such as these regarding sourcing location may be further aided by a focus on the monetary valuation of these water volumes, a subject to which Part B now turns.

Table 4.5. Baseline water stress values for stage 1 wheat sourcing regions

Country	State/region	Baseline water stress	% of durum wheat sourced from
Canada	Alberta	Too much variation	8
Canada	Saskatchewan	Too much variation	8
USA	Arizona	1.26	5
USA	California	64.10	5
USA	Montana	0.24	5
USA	North Dakota	0.32	5
Mexico	Baja California	2.44	6
Mexico	Sonora	1.18	6
Italy	Basilicata	0.71	70
Italy	Calabria	1.06	70
Italy	Campania	3.52	70
Italy	Puglia	1.25	70
France	Centre (Orleans)	0.16	10
France	Midi-Pyrenees (Toulouse)	0.19	10
France	Languedoc-Roussillon (Montpellier)	0.30	10
Greece	Western Macedonia (Kozani)	0.41	1
Greece	Central Macedonia (Thessaloniki)	0.71	1
Greece	Thessaly (Larissa)	1.10	1

Source: World Resources Institute (2013).

Part B – Unit water values along the supply chain

Having estimated the volumes of blue, green and grey water that are consumed and degraded along the pasta supply chain in Part A, Part B now turns to the monetary value of these water volumes and what this might add to water footprint assessment.

Part B is structured as follows: section 4.7 estimates the value of blue water used in pasta production; section 4.8 estimates the value of grey water, and section 4.9 comments on the suitability, in this context, of the approach to estimating green water values that was set out in Chapter Three (Part Three). As mentioned in Chapter Three, the focus here will be the direct use value that accrues to these volumes of water when they are extracted from the stream and used in agricultural, industrial and municipal settings.

Section 4.10 draws together the preceding sections and looks at the implications of the analysis. This will include a number of sensitivities which have been conducted on the values that are presented here, in part to reflect the potential of in-stream values to alter the inferences that are arrived at. Finally, section 4.11 concludes the chapter and summarises the analysis in Parts A and B.

4.7 Blue water value

The direct use values associated with blue water use in the supply chain will be considered in reverse order below, starting with the consumer use phase (stage 3) and the blue water that is consumed when cooking pasta in the home.

Consumer use phase (stage 3)

As set out at length in Chapter Three, a simple two-part formula for estimating a household demand function has been used to estimate the value of residential water use. The first part of the formula derives the value of treated water delivered to the home; the second part estimates the net consumer surplus which is equivalent to the value of raw water in the stream. The two parts of the formula are repeated directly below. In conjunction with the inputs in Table 4.6, an at site value of $8.22 (part 1) and an at source value of $1.72 (part 2), both per cubic metre, were estimated.

Part 1

$V = [(P\ x\ Q_1^{\frac{1}{E}}) \ / \ (1 - \frac{1}{E})] \ * \ [(Q_1^{1-\frac{1}{E}}) \ - (Q_2^{1-\frac{1}{e}})]$ Young and Loomis (2014)

Part 2

$CS = V - [(P_1)(Q_1 - \ Q_2)]$

Where:

E = Elasticity

P = Price

Q = Quantity

Table 4.6. Residential water value – Demand function inputs

Input	Value	Source
Q1	103.5 litres per person per day (10% reduction on Q2).	
Q2	115 litres per person per day; 42 m^3 per annum.	Environment Agency (2008)
Price (2014 USD)	6.5 (rate for highest use block 30+ m^3 in Berlin)	Global Water Intelligence (2016)
Price elasticity estimate	-0.229	Schleich & Hillenbrand (2007)
At site value (2014 USD per m^3)	8.22	
At source value (2014 USD per m^3)	1.72	

Industrial water use (stage 2)

The water used in stage 2 by the factory in Pedriganano, Italy, has been estimated with reference to the two sources set out in Chapter Three (Part Three). There it was argued that Wang and Lall (2002) and Bruneau (2007) provide the most robust estimates of the value of water *consumed* in a variety of different industries. Table 4.7 below presents the estimates that are specific to the food industry, as applicable in this context. In what follows, the average of the two values shown in Table 4.7, which is \$2.39, will be utilised. It should be noted here that only the 4 litres per kg (or 4 m^3 per tonne) that is consumed during milling and pasta processing will be valued here as the water use associated with packaging inputs is not sufficiently detailed or geographically specific.[28]

[28] It is noted here that the 4 m^3 used at stage 2 includes some unspecified quantity of water associated with energy and transportation which has been directly assigned to the product unit. However, given that it is not possible to determine how much of this 4 m^3 is accounted for by energy and transportation, and the typically small volumes of water associated with these items, it is assumed here that all of the 4 m^3 at stage 2 is used during direct pasta production and thus is subject to valuation as described here.

Table 4.7. Food industry values used in the pasta supply chain case study

Supply chain location at Stage 2 (Policy site)	Source	Method	Value type	Water volume measure	Original value m^3 (currency)	2014 USD per m^3
UK and Belgium	Wang & Lall (2002)	Production function	MV	Consumption	2.57 (Yuan)	1.87
UK and Belgium	Bruneau (2007)	Alternative cost	AV	Consumption	2.5 (CAD)	2.92
						2.39 (Average)

MV = Marginal Value. AV = Average Value.

Agricultural water use (stage 1)

Table 4.8 below presents the values that have been selected from the literature for each of the countries at stage 1 other than the USA (i.e. Canada, Mexico, Italy, France and Greece). As shown, there are only a handful of values for countries other than the USA, and in some cases, particularly outside North America, just one value is available. Table 4.9 below sets out the values that have been applied for each of the four states in the USA at stage 1 (Arizona, California, North Dakota and Montana). These are presented separately below because they are derived from averages, across numerous value estimates, which were recorded in the USA agricultural values data pool that was described in Part Two of Chapter Three. The estimates used to derive the values in Table 4.9 were short run, at site values, where the water was measured in terms of withdrawal or application and applied to crops of low or unknown value.[29] As shown, there were a number of suitable value estimates for Arizona and California, however, for Montana and North Dakota, the values assigned originate from the broader regions within which these states sit as there were no appropriate values for the states themselves.

The first thing that is noticeable about the values in Tables 4.8 and 4.9 is that, Canada and the USA aside, the literature did not provide a bespoke value for each sub-national location that water footprint data was available for in Mexico, France and Greece, and no irrigation value was available for Italy. In the case of Greece, there is nonetheless a good correspondence between study and policy sites, all being regions in the very northern part of the country. Likewise, the value for France, which is applicable to the

[29] The only exception to this were the estimates used to derive the value for North Dakota which were measured in-stream rather than at-site.

southern region, can be assumed to be appropriate for the Midi-Pyrenees and Languedoc-Roussillon regions, and arguably, given the size of France, the Centre region as well. However, vis-à-vis Mexico, no values were available for the western and northern regions of the country where Baja California and Sonora are located, so an average of those values which pertain to the central and eastern regions, has had to be assumed here. This is obviously a limitation in this context, however, the granularity of data is simply not sufficient in many locations for bespoke regional values to be differentiated and thus the analysis of Mexico, in particular, is best viewed at the national level. This will be taken into account in section 4.10 when the analysis will sensitise the unit values presented here. Finally, as mentioned, a value for irrigation water in Italy was not available. Therefore, a value has been transferred from El Chami *et al.* (2015) who have estimated the value of irrigation water *consumed* in the south east of England. This value has been chosen here because it is specific to wheat production and because the UK and Italy are both analogous advanced western European countries. However, the implication of using the value from El Chami *et al.* (2015) is that, unlike the other values in Table 4.8 which are for withdrawal or application and thus represent a lower bound estimate of water consumed, utilising a specific value for water consumed means that it will be significantly higher the other values at stage 1. As a result, whilst the value utilised for Italy will enable an approximation to be made regarding the *total* value of water consumed during stage 1, it will not be helpful when it comes to looking at the *relative* value of water in different locations.

As a result, the unit value for Italy will be omitted in the discussion of relative values, and the implications that stem from this, in what follows. Indeed, whilst every effort has been made in Table 4.8 and 4.9 to reflect a common scenario (i.e. low valued crops for which the value of irrigation water is measured at site and in the short run) so as to be able to comment on relative values in different locations, inevitably there are small variations in the exact type of value shown, variations which are magnified by the number of countries involved in this analysis (six) when compared to the tea case study (three). As result, as mentioned in the tea case study, the values in Table 4.8 and 4.9 should be considered indicative only; they would need to be investigated using fully consistent primary valuation techniques, in each location, if a policy relevant action was contingent on them.

Table 4.8. Agricultural values used in the pasta supply chain (Non – USA)

Supply chain location at Stage 1 (Policy site)	Source	Method	Value type	At site/ at source	Long run/short run	Water volume measure	Crop value	Original value m^3 (currency)	2014 USD per m^3	Study location (Study site)
Canada (Alberta)	Bruneau (2007)	Production function	AV	At site	Unclear	Application	Low (wheat)	0.14 [a] (CAD)	0.16	Saskatchewan and Alberta
Canada (Alberta)	Samarawickrema & Kulshreshtha (2008)	Yield comparison	AV	Unclear	Short	Application	Various – mostly low value	0.05 (CAD)	0.05	Four basins in Alberta [b]
AVERAGE									**0.11**	
Canada (Saskatchewan)	Bruneau (2007)	Production function	AV	At site	Unclear	Application	Low (wheat)	0.14 (CAD)	0.16	Saskatchewan and Alberta
Canada (Saskatchewan)	Samarawickrema & Kulshreshtha (2008)	Yield comparison	AV	Unclear	Short	Application	Various – mostly low value	0.05 (CAD)	0.05	Two basins in Saskatchewan [c]
Canada (Saskatchewan)	Kulshreshtha and Brown (1990)	Yield comparison	AV	Unclear	Short	Unclear	Various – mostly low value	0.06 (CAD)	0.09	Saskatchewan
AVERAGE									**0.10**	
Mexico (Sonora and Baja California)	Puente Gonzalez (2007) *in* EVRI (2011)	Opportunity cost	Unclear	Unclear	Unclear	Unclear	Low (Maize)	0.98 (MX$)	0.15	Veracruz
Mexico (Sonora and Baja California)	Arias Rojo (2007) *in* EVRI (2011)	Opportunity cost	Unclear	Unclear	Unclear	Unclear	Unclear	1.99 (MX$)	0.32	Satillo
Mexico (Sonora and Baja California)	Zetina Espinosa *et al.* (2013)	Linear Programming	MV	At site	Unclear	Unclear	Various – mostly low value	3.34 [d] (MX$)	0.5	Hidalgo
Mexico (Sonora and Baja California)	Zetina Espinosa *et al.* (2013)	Linear Programming	MV	At site	Unclear	Unclear	Various – mostly low value	0.12 [e] (MX$)	0.02	Hidalgo
AVERAGE									**0.24**	
France (Centre and Languedoc-Roussillon)	Tardieu & Prefol (2001) *in* Hussain *et al.* (2007)	Unclear	AV	Unclear	Unclear	Unclear	Unclear	0.18 (USD)	0.20	France (Adoor-Garonne basin)

Greece (Western Macedonia and Central Macedonia)	Latinopoulos *et al.* (2004)	Hedonic	MV	At site	Long	Withdrawal	Unclear	0.06 (Euros)	0.12	Greece (Chalkidiki)
Italy (Calabria and Campania)	El Chami *et al.* (2015)	Yield comparison	AV	At site	Unclear	Consumption	Low (wheat)	0.24 (GBP)	0.34	UK (East of England)

MV = Marginal Value. AV = Average Value. [a] Value is an average across two wheat types (HRS and SRS). [b] Unit value is an average across four sub-basins, within Alberta, that are part of the SSRB (South Saskatchewan River Basin). [c] Unit value is an average across two sub-basins, within Saskatchewan, that are part of the SSRB (South Saskatchewan River Basin). [d] Median value in range given for winter season. [e] Median value within range given for summer season. Values converted from local currency to 2014 USD using World Bank PPP exchange rates and Implicit Price Deflator (Appendix 3). See Chapter Three.

Table 4.9. Agricultural values used in the pasta supply chain (USA)

Supply chain location at Stage 1 (Policy site)	Geographic region used	Number of estimates in the database for specified geographic region	2014 USD per m^3
Arizona	Arizona	13	0.08
California	California	4	0.07
Montana	Census division 8	20	0.06
North Dakota	Census division 4	3	0.07

Figure 4.6 (high scenario) and Figure 4.7 (low scenario) below set out the unit values of blue water that have been assigned to each of the three stages along the pasta supply chain, together with the value of the volume of water used at each stage to produce one tonne of pasta. For both Alberta and Saskatchewan in Canada, and Mexico, an average of the values selected from the literature has been utilised (i.e. $0.10, $0.11 and $0.24 respectively).

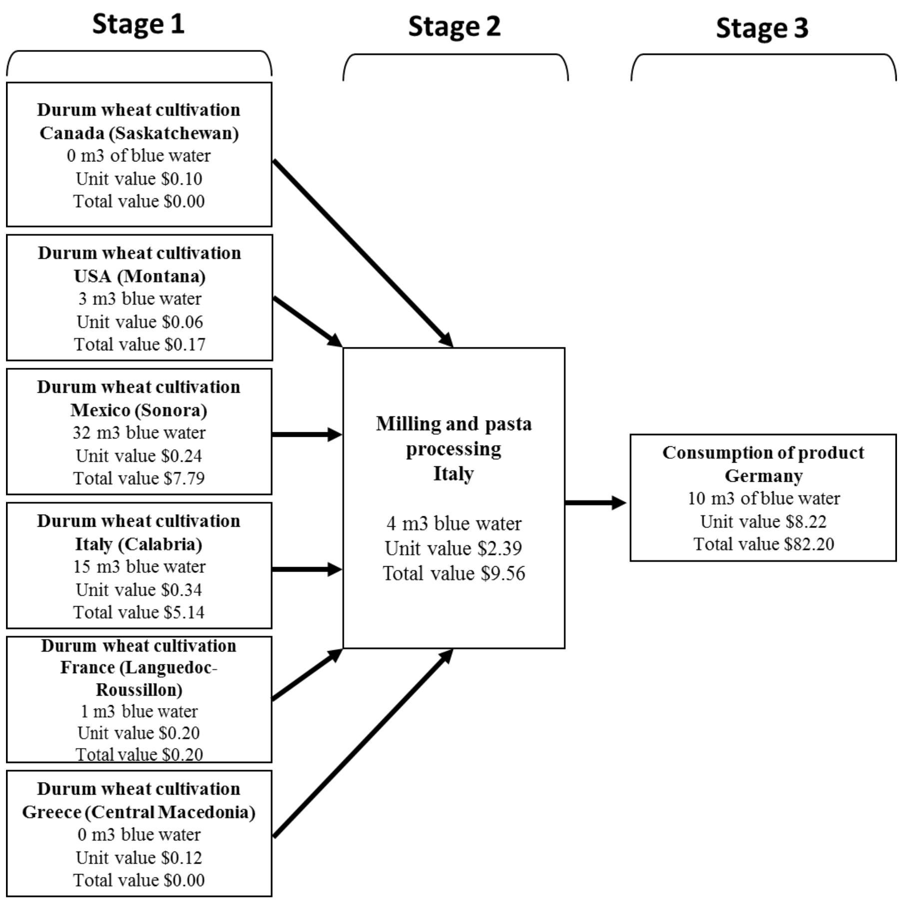

Figure 4.6. Blue water values assigned to each stage of the pasta supply chain (high scenario).

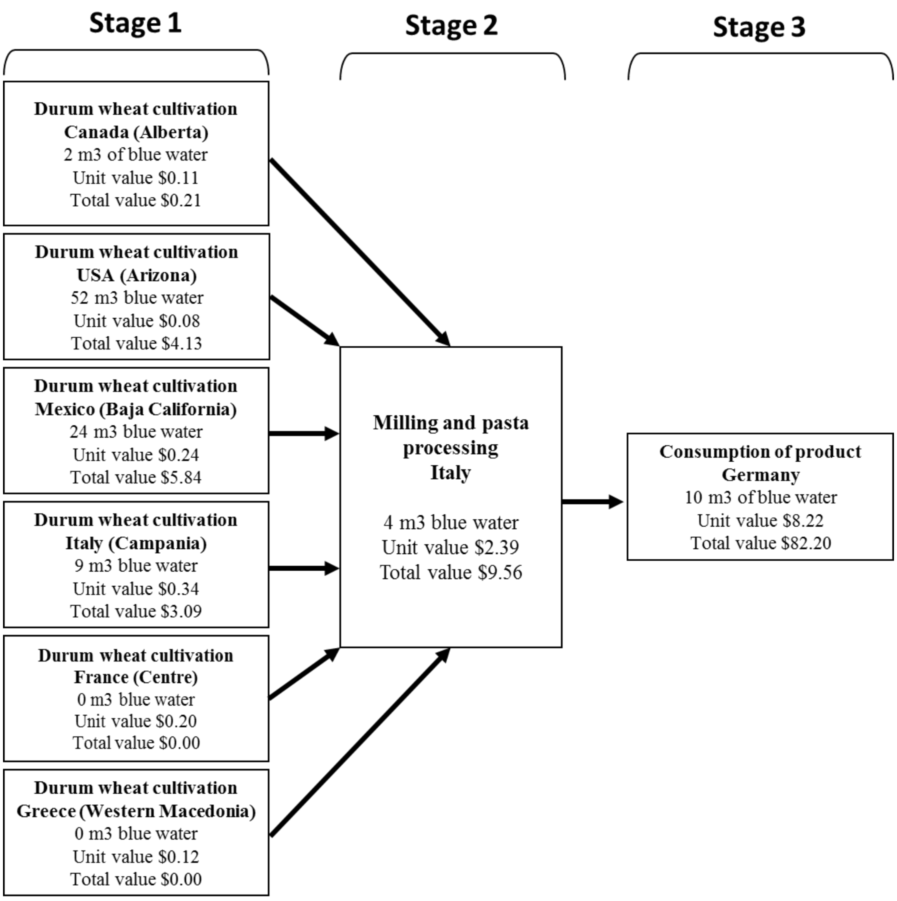

Figure 4.7. Blue water values assigned to each stage of the pasta supply chain (low scenario).

As shown in Table 4.10 below, in the high scenario, the large unit value associated with the water consumed during stage 3 ensures that whilst it accounts for only 15% by volume, it represents 78% by of the total value of blue water consumed in the production of one tonne of wheat.[30] Similarly, the value of water consumed in the factory stages is such that whilst stage 2 accounts for only 6% by volume, it is associated with 9% of the total value of the water consumed. Of the countries at stage 1, the relatively high unit value assigned to Italy (reflecting as referred to previously the value of water consumed rather than withdrawn/applied) ensures that whilst it accounts for 29% by total volume at stage 1, it represents 39% of the total value of water consumed at stage 1. Indeed, even though the presence of a dissimilar value for Italy distorts the picture somewhat, the relatively high unit value in Mexico sees only a slight imbalance between volume (63%) and value (59%) at stage 1. By comparison to Mexico and Italy, the relatively low

[30] This may have been slightly less if values for irrigation water at stage 1 had been available for water *consumed* in Canada, USA, Mexico, France and Greece.

unit value in the USA produces a noticeable imbalance in volume (6%) and value (1%). The total value of blue water consumed in the high scenario is approximately $105 per tonne of pasta, or, using the prevailing nominal exchange rate in mid 2017 (1 USD = 0.77 GBP), £81.

Table 4.10. Blue water value and volume distribution in the pasta supply chain (high scenario)

Stage (location)	Volume of blue water (m^3)	Unit value (USD 2014)	Value of blue water consumed (USD 2014)	% of total blue water volume	% of total blue water value	% of stage 1 volume	% of stage 1 value
1 (Canada Saskatchewan)	0	0.10	0	0	0	0	0
1 (USA – Montana)	3	0.06	0.17	5	<1	6	1
1 (Mexico – Sonora)	32	0.24	7.79	49	7	63	59
1 (Italy – Calabria	15	0.34	5.14	23	5	29	39
1 (France Languedoc-Roussillon)	1	0.20	0.20	2	<1	2	2
1 (Greece – Central Macedonia)	0	0.12	0	0	0	0	0
2 (Italy)	4	2.39	9.56	6	9		
3 (Germany)	10	8.22	82.20	15	78		
Total	**65**		**105.07**	**100**	**100**	**100**	**100**

The observations noted for the high scenario are replicated in the low scenario set out in Table 4.11 below. However, in the low scenario the relatively large unit value associated with Mexico is far clearer, becoming apparent in the imbalance between volume (28%) and value (44%) at stage 1. Conversely, the low unit value in the USA, this time in Arizona as opposed to Montana, ensures that there is a disparity between volume (60%) and value (31%), albeit this time the other way around. The total value of blue water consumed in the low scenario is also approximately $105 per tonne of pasta, or, £81.

Table 4.11. Blue water value and volume distribution in the pasta supply chain (low scenario)

Stage (location)	Volume of blue water (m^3)	Unit value (USD 2014)	Value of blue water consumed (USD 2014)	% of total blue water volume	% of total blue water value	% of stage 1 volume	% of stage 1 value
1 (Canada Alberta)	2	0.11	0.21	2	<1	2	2
1 (USA – Arizona)	52	0.08	4.13	51	4	60	31
1 (Mexico – Baja California)	24	0.24	5.84	24	6	28	44
1 (Italy – Campania	9	0.34	3.09	9	3	10	23
1 (France Centre)	0	0.20	0	0	0	0	0
1 (Greece – Western Macedonia)	0	0.12	0	0	0	0	0
2 (Italy)	4	2.39	9.56	4	9		
3 (Germany)	10	8.22	82.20	10	78		
Total	**101**		**105.04**	**100**	**100**	**100**	**100**

4.8 Grey water value

As referred to previously in Chapter Three, it is assumed here that the unit value of grey water degradation is equal to the unit value of blue water consumption. This assumption has been made because grey water refers to the volume of blue water that is necessary to assimilate or abate pollution. As we have seen, blue water consumption impacts a variety of in-stream ESS (waste assimilation, wildlife habitat and recreation) and off-stream extractive uses. However, only the values associated with of off-stream extractive uses are available here so the *unit* values of grey water are identical to those presented in the previous section. Figures 4.8 and 4.9 below re-state the applicable unit values, and set out the value of grey water along the supply chain based on these unit value estimates, for both the high and low scenarios.

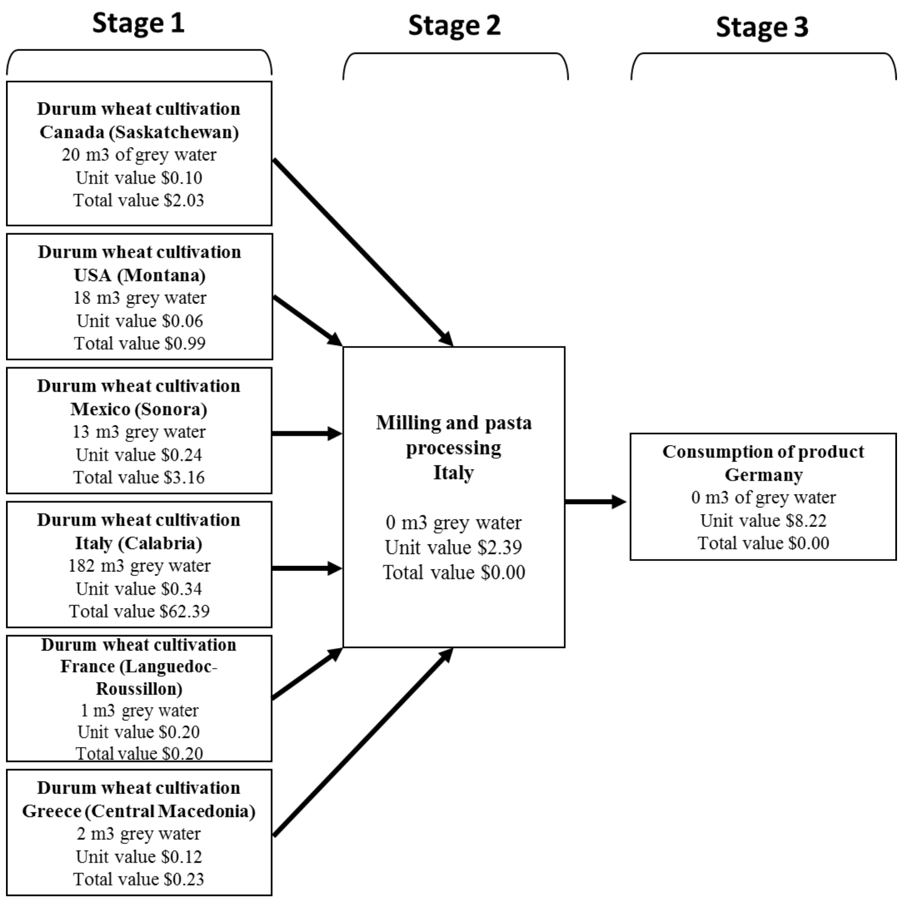

Figure 4.8. Grey water values assigned to each stage of the pasta supply chain (high scenario).

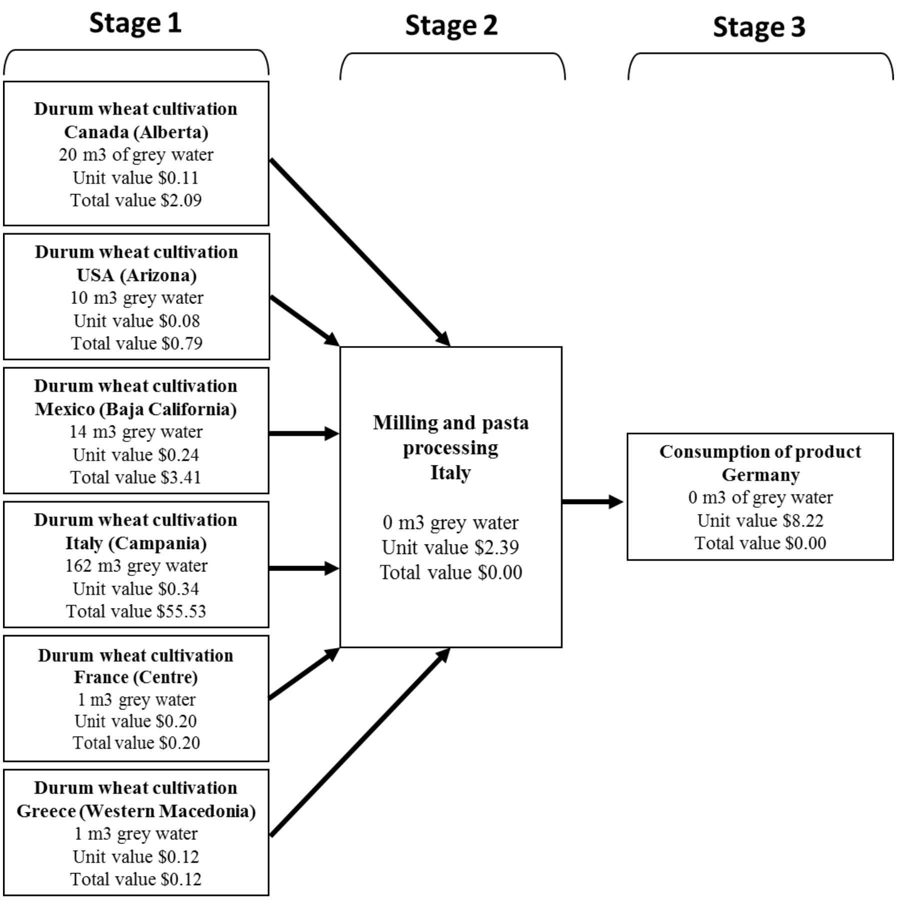

Figure 4.9. Grey water values assigned to each stage of the pasta supply chain (low scenario).

Table 4.12 below sets out the total value of grey water in the high scenario. Again, by comparison to the other countries, the inflated unit value of water in Italy is apparent in Table 4.12 with grey water representing 77% of total volume but 90% of total value. In addition, the comparatively low value that prevails in Montana in the USA ensures that this region accounts for 8% of the total volume of grey water used to produce a tonne of pasta, but only 1% of the total value. It is a similar picture in Saskatchewan, Canada. There is no grey water associated with stages 2 and 3. The total value of grey water in the high scenario is $69 per tonne of pasta, or, £53.

Table 4.12. Grey water value and volume distribution in the pasta supply chain (high scenario)

Stage (location)	Volume of grey water (m^3)	Unit value (USD 2014)	Value of grey water degraded (USD 2014)	% of total grey water volume	% of total grey water value
1 (Canada Saskatchewan)	20	0.10	2.03	8	3
1 (USA – Montana)	18	0.06	0.99	8	1
1 (Mexico – Sonora)	13	0.24	3.16	6	5
1 (Italy – Calabria)	182	0.34	62.39	77	90
1 (France Languedoc-Roussillon)	1	0.20	0.20	<1	<1
1 (Greece – Central Macedonia)	2	0.12	0.23	1	1
Total	**236**		**69.01**	**100**	

Table 4.13 below sets out value of grey water in the low scenario which reflects similar relative relationships to those noted in the high scenario. The total value of grey water in the low scenario is $62 per tonne of pasta, or, approximately £48.

Table 4.13. Grey water value and volume distribution in the pasta supply chain (low scenario)

Stage (location)	Volume of grey water (m^3)	Unit value (USD 2014)	Value of grey water degraded (USD 2014)	% of total grey water volume	% of total grey water value
1 (Canada Alberta)	20	0.11	2.09	10	3
1 (USA – Arizona)	10	0.08	0.79	5	1
1 (Mexico – Baja California)	14	0.24	3.41	7	5
1 (Italy – Campania	162	0.34	55.53	78	89
1 (France Centre)	1	0.20	0.20	<1	<1
1 (Greece – Western Macedonia)	1	0.12	0.12	<1	<1
Total	**208**		**62.15**	**100**	**100**

4.9 Green water value

Part Three of Chapter Three set out the approach to valuing green water in light of the available valuation data collected during this study. By way of a recap, green water in this context is not rain water as such but the water that is evapotranspired by the potato crop during its growth phases, or, in other words, it is the volume of water that is usefully absorbed by the crop. As such, it had been anticipated that values for irrigation water *consumed* by the crop would be used as a proxy for the value of green water. However, apart from the value applied in Italy, these were not available in the supply chain locations in stage 1, and as a result, the value of green water will be assumed to be equivalent to the *at source* value of artificially applied irrigation water.[31] In order to estimate at source values, the difference between the mean and median at site and at source values for irrigation water in the USA and ROW value databases, as a whole, was assessed. The largest difference (USA database; mean value) showed that at source values were typically 60% of at site values; the smallest difference (ROW database; median value) showed that at source values were typically 80% of at site values. As a result, these two measures were used to deflate the at site blue water values used above to provide an estimate of the at source value at each stage 1 location. Sensitivity 1 below (or S1) reflects the at source value at 60% of the at site value; sensitivity 2 (or S2) reflects 80%. In many ways this is a crude estimate of the value of green water. However, as mentioned earlier, Aldaya *et al.* (2010a) points to the contemporary significance of green water in the international trade in crops, and thus ensuring that the value of green water is incorporated here in some way, is important. What is more, by using a measure of the at source value of water diverted or applied, this is in many ways a conservative estimate of the value of water that is consumed, and thus becomes more defensible.

Figures 4.10 and 4.11 below present the unit values of green water, together with the value of the green water consumed at *each stage* of the supply chain, for both low and high scenarios. There is no green water consumed in stages 2 and 3 of the supply chain.

[31] As with the other unit values utilised at stage 1, the value for Italy has likewise been adjusted as described above in order to reflect the *at source* value of the consumed water.

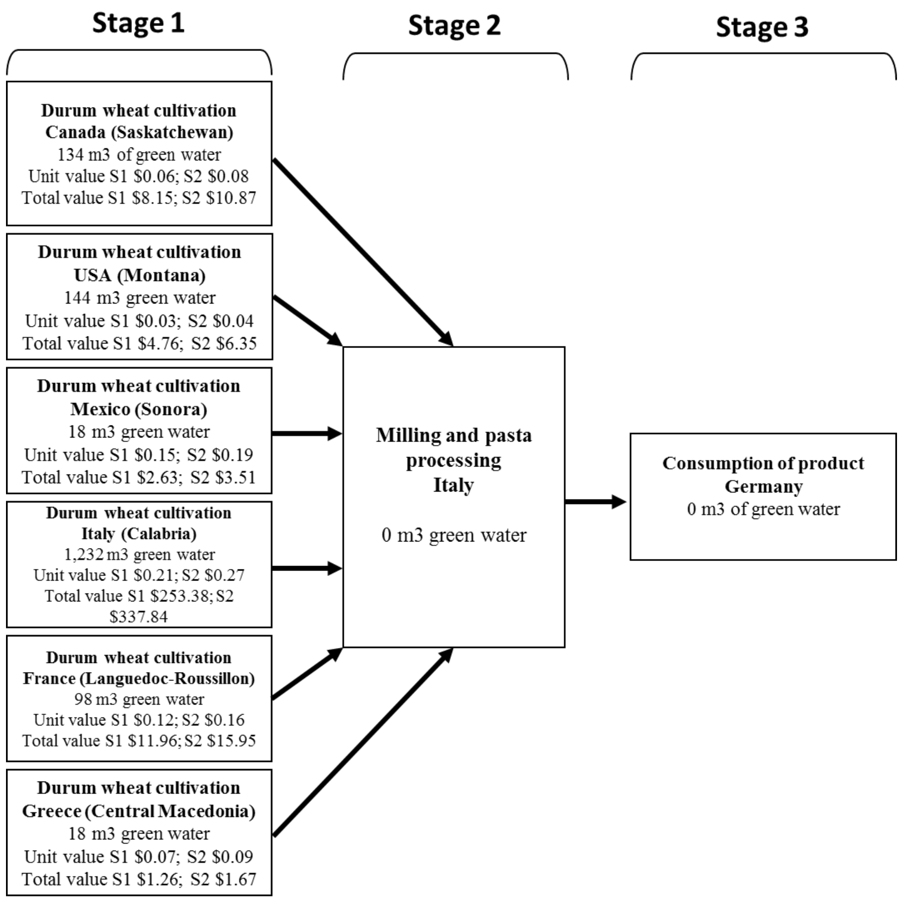

Figure 4.10. Green water values assigned to each stage of the pasta supply chain (high scenario).

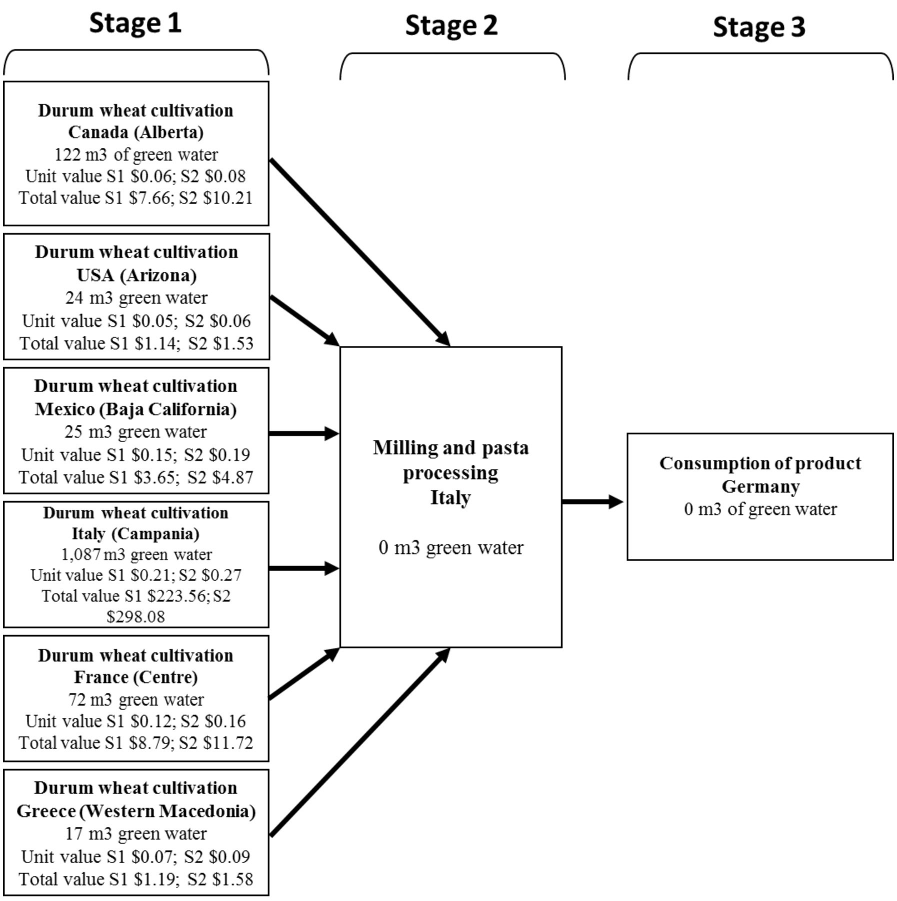

Figure 4.11. Green water values assigned to each stage of the pasta supply chain (low scenario).

Tables 4.14 and 4.15 below present the *total* value of green water in the low and high scenarios. From these tables it is clear that the value of green water associated with the quantity of wheat used in a tonne of pasta, in both the low and high scenarios, is far greater than the value of the wheat crop itself. The price of a tonne of wheat on the world market in 2017 is between $130 and $150 per tonne (IMF, no date). Given that approximately 1.4 tonnes of wheat are associated with a tonne of pasta (1 tonne of wheat divided by the product fraction 0.72) and that the lowest estimate of green water in Tables 4.14 and 4.15 was $246, then assigning a value of $176 ($246/1.4) clearly does not stand scrutiny. Indeed, even ignoring of the presence of blue water values, a farmer would clearly not be willing to pay for water at these levels, and ultimately, no matter what the valuation method employed, the value of water in agriculture is a derived demand and driven by the crop price. As a result of this analysis, and also that presented in the other two case studies, the value of green water has been excluded here and the approach to valuing green water will be revisited in Chapter Seven when the conclusions and recommendations from the project as a whole are presented.

Table 4.14. Green water value and volume distribution in the pasta supply chain (high scenario)

Stage (location)	Volume of Green water (m^3)	Unit value S1 (USD 2014)	Unit value S2 (USD 2014)	Value of green water consumed S1 (USD 2014)	Value of green water consumed S2 (USD 2014)	% of total green water volume	% of total green water value
1 (Canada Saskatchewan)	134	0.06	0.08	8.15	10.87	8	3
1 (USA – Montana)	144	0.03	0.04	4.76	6.35	9	2
1 (Mexico – Sonora)	18	0.15	0.19	2.63	3.51	1	1
1 (Italy – Calabria	1,232	0.21	0.27	253.38	337.84	75	90
1 (France Languedoc-Roussillon)	98	0.12	0.16	11.96	15.95	6	4
1 (Greece – Central Macedonia)	18	0.07	0.09	1.26	1.67	1	<1
Total	**1,644**			**282.15**	**376.20**	**100**	**100**

Table 4.15. Green water value and volume distribution in the pasta supply chain (low scenario)

Stage (location)	Volume of Green water (m^3)	Unit value S1 (USD 2014)	Unit value S2 (USD 2014)	Value of green water consumed S1 (USD 2014)	Value of green water consumed S2 (USD 2014)	% of total green water volume	% of total green water value
1 (Canada Alberta)	122	0.06	0.08	7.66	10.21	9	3
1 (USA – Arizona)	24	0.05	0.06	1.14	1.53	2	<1
1 (Mexico – Baja California)	25	0.15	0.19	3.65	4.87	2	1
1 (Italy – Campania	1,087	0.21	0.27	223.56	298.08	81	91
1 (France Centre)	72	0.12	0.16	8.79	11.72	5	4
1 (Greece – Western Macedonia)	17	0.07	0.09	1.19	1.58	1	<1
Total	**1,347**			**245.99**	**327.99**	**100**	**100**

4.10 Implications

Tables 4.16 and 4.17 below present the total value of the blue and grey water associated with the production of one tonne of potato crisps in the high and low scenarios. As mentioned, this only encompasses the direct use value of these water volumes, and it only includes those aspects of water use within the supply chain that were geographically and functionally specific i.e. it excludes the water associated with packaging inputs. As depicted, the total direct use value of the water footprint varies between $167 per tonne of pasta in the low scenario and $174 per tonne of pasta in the high scenario. Given that these values are no longer in evidence when the water is consumed or degraded, they effectively represent costs, and therefore, as modelled, sourcing from the combination of countries in the low scenario is preferable to the combination in the high scenario. This conclusion may appear marginal as the difference between the two scenarios is only approximately $7 per tonne in spite of the fact that as we have seen, in volumetric terms, there is a 300 cubic metre difference. Nonetheless, when multiplied over the millions of tonnes of pasta which are consumed, this monetary figure, much of which may not already be internalised, becomes important. This will be discussed at greater length below, and during Chapter Seven when the conclusions from the project as a whole are presented.

Table 4.16. Total value of the blue and grey water used to produce one tonne of pasta (high scenario)

Water footprint component	Value USD 2014	Value GBP
Blue	105.07	80.90
Grey	69.01	53.14
Total value	174.08	134.04

Table 4.17. Total value of the blue and grey water used to produce one tonne of pasta (low scenario)

Water footprint component	Value USD 2014	Value GBP
Blue	105.04	80.88
Grey	62.15	47.86
Total value	167.19	128.74

Tables 4.18 and 4.19 below set out how the total value of blue and grey water breaks down by supply chain stage in both the high and low scenarios, thus directly addressing RQ2. As shown, in both scenarios, it is the values associated with the blue water used in the consumption of pasta in Germany, and grey water that is a by-product of wheat cultivated in Italy, that account for the largest shares of total blue and grey water value.

However, as mentioned previously, the blue and grey water unit value used for Italy was not directly comparable with the unit values applied to the other countries at stage 1 given that it was applicable to water consumption. As a result, the conclusion regarding the relative value of grey water degraded in Italy should be treated with some caution. However, it is representative of the fact that 70% of durum wheat is sourced from Italy.

Table 4.18. Total value breakdown by supply chain stage (high scenario)

Stage (location)	% of total value of blue and grey water
1 (Canada Saskatchewan) Blue water	0.0
1 (Canada Saskatchewan) Grey water	1.2
1 (USA – Montana) Blue water	0.1
1 (USA – Montana) Grey water	0.6
1 (Mexico – Sonora) Blue water	4.5
1 (Mexico – Sonora) Grey water	1.8
1 (Italy – Calabria Blue water	3.0
1 (Italy – Calabria Grey water	35.8
1 (France Languedoc-Roussillon) Blue water	0.1
1 (France Languedoc-Roussillon) Grey water	0.1
1 (Greece – Central Macedonia) Blue water	0.0
1 (Greece – Central Macedonia) Grey water	0.1
2 (Italy) Blue water	5.5
3 (Germany) Blue water	47.2
Total	**100**

Table 4.19. Total value breakdown by supply chain stage (low scenario)

Stage (location)	% of total value of blue and grey water
1 (Canada Alberta) Blue water	0.1
1 (Canada Alberta) Grey water	1.3
1 (USA – Arizona) Blue water	2.5
1 (USA – Arizona) Grey water	0.5
1 (Mexico – Baja California) Blue water	3.5
1 (Mexico – Baja California) Grey water	2.0
1 (Italy – Campania) Blue water	1.8
1 (Italy – Campania) Grey water	33.2
1 (France Centre) Blue water	0.0
1 (France Centre) Grey water	0.1
1 (Greece – Western Macedonia) Blue water	0.0
1 (Greece – Western Macedonia) Grey water	0.1
2 (Italy) Blue water	5.7
3 (Germany) Blue water	49.2
Total	**100**

The values thus far presented for each stage 1 sourcing location for wheat have been based on the blend of wheat sources utilised at stage 1 (see Figure 4.1). However, to judge the optimum sourcing location from a value perspective, what is needed is to understand the value of the blue water consumed, and grey water degraded, in the cultivation of a common quantity of wheat. Table 4.20 below presents the total value of the blue and grey water associated with producing a tonne of wheat in each stage 1 sourcing location. As shown, this includes all of the sub-regions and states presented in Table 4.1 and not just those that fell within the high and low scenarios commented on already. However, Table 4.20 does *not* include the value of a tonne of wheat sourced from the four Italian regions given that, as mentioned, the unit value for Italy is not directly comparable with the unit values used for other locations at stage 1.

What is quite clear in Table 4.20 is that there is substantial variation in terms of what a farmer might be willing to pay for the irrigation water used to produce durum wheat across the 14 locations. Indeed, whilst Table 4.20 presents the best available data in terms of volume and values, it does suggest that farmers in the various locations are either facing different costs levels, or that they are able to realise different prices locally for their crop, which may in turn be dependent upon whether and to what extent it is irrigated and the quality differences that this may bring. Broadly speaking, it is noticeable that, as expected, the highest unit values are in evidence in the locations which use the lowest quantity of blue and grey water. However, the exception to this is Mexico which experiences both relatively large unit values and relatively high levels of blue and grey water use. As a result, the values estimated for Mexico should be treated with some caution here.

Table 4.20 also suggests that, in terms of the unit value of a metre cubed of irrigation water, it is the four states in the USA, and to a lesser extent to the two Canadian regions, which represent the optimum sourcing location. However, when the prevailing unit value in each location is used to estimate the *volume adjusted* value in each location, it is clear that the three French regions impose the lowest costs *per tonne* of wheat, even though at $0.20 per cubic metre, the unit value is the second highest presented. Indeed, in light of this, from the perspective of volume adjusted value, it clear that France potentially represents the optimum wheat source for the company, whilst Mexico, with its high unit value together with large volumes of blue and grey water, is the least optimum wheat sourcing location. This conclusion accords with the volumetric perspective regarding the optimum sourcing location as France consumes and pollutes the lowest volume of blue and grey water (Table 4.1). However, it contradicts the volumetric perspective when choosing the least optimum sourcing location (Montana), which clearly shows the merit of going beyond the approach taken by Ruini *et al.* (2013)

Table 4.20. Value of blue and grey water uses to produce one tonne of wheat in each location

Location	Blue water (m^3)	Grey water (m^3)	Unit vale (USD 2014)	Total value of blue water (USD 2014)	Total value of grey water (USD 2014)	Total value of blue and grey water (USD 2014)
Orleans	2	6	0.20	0.41	1.22	1.63
Toulouse	4	6	0.20	0.81	1.22	2.03
Montpellier	4	7	0.20	0.81	1.42	2.24
North Dakota	1	184	0.07	0.07	12.86	12.93
Kozani	2	119	0.12	0.23	13.84	14.07
Montana	53	299	0.06	2.92	16.48	19.41
Larissa	34	139	0.12	3.95	16.16	20.12
Thessaloniki	35	141	0.12	4.07	16.40	20.47
Saskatchewan	1	206	0.10	0.10	20.89	20.99
Alberta	16	202	0.10	1.67	21.14	22.82
California	522	158	0.07	36.45	11.03	47.49
Arizona	848	156	0.08	67.41	12.40	79.82
Baja California	325	186	0.24	79.12	45.28	124.40
Sonora	432	184	0.24	105.17	44.79	149.96

and taking into account values as well as volumes, and in addition highlights the relevance of RQ3. Indeed, as shown in Table 4.20, there is a potential cost saving of \$17.78 attached to sourcing one tonne of wheat from Orleans as opposed to Montana, or \$148.33 when compared to Sonora. Moreover, by having water value in monetary terms, this ensures that additional factors become relevant such as relative exchange rate fluctuations, and the costs, values and resulting trade-offs, associated with other inputs into production. These are all considerations which are beyond volume focused assessments such as Ruini *et al.* (2013).

However, these conclusions are based on what is, in some cases, limited evidence on the unit values which prevail in each geography. As a result, the standard convergent validity techniques that would usually be applied here to estimate transfer error in each location are not feasible. Therefore, given the sensitivity of the conclusions to the precise unit values applied in each location, and the importance of the *relative* differences in unit values between locations, we now move on the sensitivity analysis in order to ascertain the degree of certainty around the conclusions drawn thus far.

4.11 Sensitivity analysis

Two sensitivities will be deployed here. The first will look at the lowest unit value in evidence – Montana – and estimate the increases that would be necessary to this unit value in order for it to be comparable with the other value estimates. The second will look at the volume adjusted values set out in Table 4.20 above and estimate the increases in value that would be necessary in the lowest valued location – Orleans – for it to be comparable with the other volume adjusted values. As part of this second sensitivity, the likelihood that in-stream values could account for any increase in values will also be commented on.

Sensitivity 1

Figure 4.12 below presents the increases (right hand column) that would necessary for the lowest unit value in Montana to be comparable with the remaining unit values in each location. For example, there would need to be a 342% increase in the unit value in Montana for it to be aligned with the value in Mexico. This is based on the standard formula for estimating transfer error set out below, the difference being that the observed and transferred values refer to separate locations (Czajkowski and Scasny, 2010):

$$E_{TR} = \frac{WTP_{transferred} - WTP_{observed}}{WTP_{observed}}$$

In addition, Figure 4.12 below also presents the decreases (left hand column) that would be necessary for the largest unit value in Mexico to be comparable with the remaining unit values. For example, the unit value would need to fall by 77% to be comparable with the value in Montana.

Seeing as Czajkowski and Scasny (2010) suggest that most transfer errors are in the 0-200% range, whilst the indications are that from a unit value standpoint the states in the USA, and Canadian provinces, represent the optimum sourcing locations, one clear conclusion seems to be that Mexico (342% increase) and to a lesser extent, France (269% increase), are the least optimal locations from a unit value perspective.

% decrease	Unit value decrease	Country	USD 2014	Unit value increase	% increase
-77%	-0.19	Mexico	0.24	0.19	342%
-73%	-0.15	France	0.20	0.15	269%
-53%	-0.06	Greece	0.12	0.06	111%
-47%	-0.05	Alberta	0.10	0.05	90%
-46%	-0.05	Saskatchewan	0.10	0.05	84%
-31%	-0.02	Arizona	0.08	0.02	44%
-21%	-0.01	North Dakota	0.07	0.01	27%
-21%	-0.01	California	0.07	0.01	27%
		Montana	0.06		

Figure 4.12. Unit value sensitivities/transfer errors (1).

Beyond this, it is also clear from Figure 4.13 below, which presents the percentage increases or decreases that would be necessary for a unit value to be comparable with the unit value closest to it, that each unit value is very sensitive to even small changes. For example, it would only take a 27% increase for the unit value in Montana to be comparable with California, or a 3% increase in the unit value in Saskatchewan for it be comparable with the unit value in Alberta. As a result of this, conclusions which look to go beyond stating what, based on the evidence, looks to be the most and least preferable locations, should be treated with great caution.

% decrease	Unit value decrease	Country	USD 2014	Unit value increase	% increase
		Mexico	0.24		
-16%	-0.04			0.04	20%
		France	0.20		
-43%	-0.09			0.09	75%
		Greece	0.12		
-10%	-0.01			0.01	11%
		Alberta	0.10		
-3%	0.00			0.00	3%
		Saskatchewan	0.10		
-22%	-0.02			0.02	28%
		Arizona	0.08		
-12%	-0.01			0.01	14%
		North Dakota	0.07		
0%	0.00			0.00	0%
		California	0.07		
-21%	-0.01			0.01	27%
		Montana	0.06		

Figure 4.13. Unit value sensitivities/transfer errors (2).

Sensitivity 2

Sensitivity 2 looks at how much the 8 m^3 of blue and grey water used in Orleans (the location with the lowest volume adjusted value) would have to increase by to be comparable with the other 13 locations analysed here. Table 4.21, which is derived from Table 4.20 above, presents the difference in volume adjusted value between Orleans and each of the other locations (column two). Dividing this difference by the 8 m^3 provides an indication of how much each of the eight cubic metres would need to increase in value by (column 3) to be comparable with each of the other locations. As can be seen, given the low levels of blue and grey water use in Orleans (and France more broadly), the value of each cubic metre would have to increase to a large extent before it would become comparable with alternatives geographies outside France. Thus, based on volume adjusted values, it seems safe to conclude that France, with its low levels of blue and grey water use, represents the optimum sourcing location from a volume adjusted value perspective.

Table 4.21. Sensitivity two – unit value increases in Orleans

Location	Difference in total value of blue and grey when compare to Orleans (USD 2014)	Increase in unit value of 8 m^3 of blue and grey (USD 2014)	% increase in $0.20 unit value
Toulouse	0.41	0.05	25%
Montpellier	0.61	0.08	38%
North Dakota	11.30	1.41	694%
Kozani	12.44	1.56	764%
Montana	17.78	2.22	1,092%
Larissa	18.49	2.31	1,136%
Thessaloniki	18.84	2.35	1,157%
Saskatchewan	19.36	2.42	1,189%
Alberta	21.19	2.65	1,302%
California	45.86	5.73	2,817%
Arizona	78.19	9.77	4,803%
Baja California	122.77	15.35	7,542%
Sonora	148.33	18.54	9,112%

In addition, the requisite unit value increases (column 3) can be compared with the instream value scale presented in the previous chapter. Based on the minimum, median and maximum combined waste assimilation, wildlife habitat and recreation values that were evident in the USA being present in one location (the only country which recorded these values), the instream value scale can be adjusted for relative incomes in France using the formula set out by Czajkowski and Scasny (2010) which assumes an income elasticity of one:

$$WTP_{ps} = WTP_{ss}\left(\frac{\overline{I_{ps}}}{\overline{I_{ss}}}\right)\varepsilon$$

where WTPss is willingness to pay at the study site, WTPps is the willingness to pay estimate transferred to the policy site, and Iss and Ips are mean income levels at the study and policy sites. ϵ represents the income elasticity of willingness to pay between the mean income levels at the study and policy sites (Czajkowski and Scasny, 2010). The income data in Table 4.22 below has been used to make this adjustment and the resulting in-stream value scale for France is set out in Table 4.23.

Table 4.22. Relative income levels in France

Country	GNI Per Capita [a]	% of USA GNI Per Capita
USA	52,308.38	100
France	36,628.78	70

[a] Data sourced from UNDP (2014).

Table 4.23. In-stream value scale France (USD 2014 per m3)

Low		Median		High
0.0004	→	0.04	→	0.43

As shown, it is quite clear that the necessary increases in value of the 8 m^3 of blue and grey water used in Orleans are far in excess of the equivalent highest in-stream values in the USA.[32] Indeed, given as mentioned that the in-stream values in the USA are for the most arid parts of the country, it seems reasonable to conclude that the presence of in-stream values in France is unlikely to alter the conclusion that Orleans (or France more generally) represents the optimum sourcing location from a volume adjusted value perspective. Moreover, in-stream values will also be present to unknown and varying degrees in the other 13 locations which, held constant in this analysis, would only widen the gulf between Orleans and each location further and thereby require the presence of even greater in-stream values in Orleans.

4.12 Conclusion

In conclusion, in Part A we saw that, depending on the scenario, 98% or 99% of the water footprint associated with durum wheat pasta is associated with the durum wheat itself. Moreover, it was apparent that sourcing durum wheat from the combination of countries/regions in the low scenario produced a saving of approximately 300 m^3 per tonne of pasta. Going beyond volumes alone, it was shown that, based on considerations of water stress, the four Italian regions, together with Baja California and Sonora in Mexico, and Arizona and California in the USA, represent potential hotspots. In Part B, the total value of the blue and grey water associated with a tonne of pasta was estimated as varying between $167 in the low scenario, and $174 in the high scenario, despite the latter accounting for an additional 300 m^3 per tonne of pasta thus highlighting the importance of values as well as volumes. In addition, based on unit values alone, it was suggested that Montana was the optimum sourcing location, although this was found to be very sensitive to even small changes in unit values. What was clear from a unit value perspective was that Mexico and France appeared to be the least favourable sourcing locations. However, when volume adjusted values were taken in to account, in the

[32] As noted in Chapter Three, in-stream ESS values are additional to agricultural values which are net of extraction costs (i.e. the agricultural value is at source). However, given that at source agricultural values were not available here, the in-stream value scale is applied to at site agricultural values on the assumption of minimal/similar extraction costs across stage 1 sourcing locations.

context of a common quantity of durum wheat, it was shown that despite the large unit value in France, it (and Orleans in particular) represents the optimum sourcing location given the low volumes of blue and grey water used. Moreover, this conclusion was found to hold even in the face of substantial increases in unit values, which it was concluded, were unlikely to be associated with the presence of in-stream ESS in France. In line with the conclusions regarding unit values considered in isolation, in volume adjusted terms, Mexico was again found to be the least favourable sourcing location (shortly followed by Arizona and California). However, this was shown to contradict the volumetric perspective and highlighted the importance of taking monetary values into account, as did the relative cost savings associated with sourcing from alternative locations. Furthermore, it was suggested that these monetary values bring other factors into the analysis such as relative exchange rates and the costs, values and ensuing trade-offs associated with other inputs into production. Again, all of these considerations are beyond volume-focused analyses such as Ruini *et al.* (2013).

These conclusions are broadly in line with the analysis of hotspots. Nonetheless, all of these conclusions were reached without the inclusion of green water in the analysis which it was argued could not be valued in the way anticipated. Indeed, it must be stressed here, particularly given the number and range of regions/countries considered at stage 1, that this case study perhaps represents the extent of what is possible with the method set out in this thesis. Whilst each value presented represents the best that is available in the literature at the present time, dissimilar numbers of value estimates in each location, and small differences in the type of value itself (although we have tried to be explicit about these throughout), together with the sensitivity of the conclusions to the exact values applied, inevitably mean that the *relative* differences in unit values that are crucial in this context would need to be tested thoroughly with consistent valuation techniques if decision relevant values were required. Moreover, the results also indicate very different willingness to pay by farmers in each of the sourcing locations, which again may suggest that they are facing different costs and prices, but is also further reason for additional analysis if decision relevant values are required.

Having examined the volumes, and monetary values, associated with the pasta supply chain, we now turn to the second case study – the tea supply chain – which is presented in Chapter Five.

5. The tea supply chain

This chapter sets out the tea supply chain case study, volumetric water data and supporting information for which, has been obtained from secondary sources as detailed below.

Part A summarises the tea water footprint i.e. the volumes of green and blue consumptive water use, and degradative grey water use, at each point along the supply chain. Part B summarises the attendant monetary values that have been assigned to these volumes of water based on the approach set out in Chapter Three (Part Three).

Part A – The tea water footprint

Part A begins by setting out the production unit that is the subject of analysis in this chapter, and providing an overview of the associated supply chain map (section 5.1). Section 5.2 sets out the consumptive blue and green water use, and degradative grey water burden, for the *supply chain* water footprint directly associated with inputs. Section 5.3 repeats this for the *operational* water footprint directly associated with inputs. Section 5.4 then describes the supply chain and operational *overhead* water footprints, before section 5.5 details the assumptions made regarding consumptive blue water use during the consumer use phase (i.e. the water used by the end consumer when drinking tea). Section 5.6 details those aspects of the analysis which are out of scope. Finally, section 5.7 summarises the total water footprint of black tea.

5.1 Product units and supply chain map

This case study draws on Jeffries *et al.* (2012) who examined the water footprint associated with one box containing 50 grams of black tea, with additional assumptions as detailed below. However, as mentioned in the previous chapter, given that larger production quantities tend to be more meaningful units of analysis when the emphasis is on monetary values (which only tend to register in volumes which exceed those associated with individual products), the water footprint associated with one tonne of black tea will also be considered. This will aid the comparison, on a like for like basis, with the pasta and potato crisp case studies, both of which have also been estimated at the one tonne level. The one tonne scenario is based on multiple (20,000) 50g boxes i.e. linear aggregation is assumed here. Whilst it is acknowledged that there may be some economies of scale associated with larger production quantities, there will also be water use associated with additional packaging and palletisation. It is therefore assumed that the overall effect is a zero-sum outcome.

The key stages in the production of tea, together with their geographical location, are set out in the supply chain map shown in Figure 5.1 below.

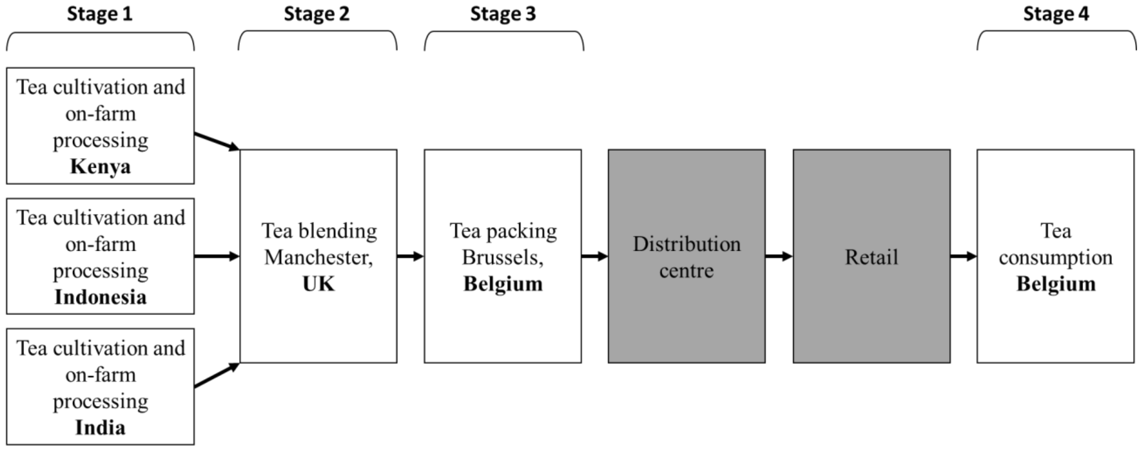

Figure 5.1. Tea supply chain map. Stages in grey are excluded from analysis of the water footprint. Adapted from Jeffries *et al.* (2012).

Crop cultivation (stage 1) occurs in the Rift Valley (Kericho) and Central Highlands (Nyeri) of Kenya, in the Jawa Barat province of Indonesia (Agrabinta), and in the state of Tamil Nadu in southern India (Kotagiri and Coonor). As shown in Table 5.1 below – which sets out the top 15 tea producing countries in 2013 together with the associated *country average* water footprint – each of these three countries reside in the top 10 global tea producing nations. Following stage 1, the tea is first sent to the UK (Manchester) for blending (stage 2), before it is packed (stage 3) in Belgium (Brussels). Final consumption of the tea by the end consumer (stage 4) is also assumed to occur in Brussels.

It should be noted here that Jeffries *et al.* (2012) excluded grey water in their estimation of the tea water footprint. This was because in their study, which was a comparative analysis between WFA and LCA, the latter appears to have been unable to address water quality issues in a way that fell within the scope of the work, and thus grey water was excluded altogether. Given this, as will be detailed in what follows, where possible the data in Jeffries *et al.* (2012) has been supplemented with data from the *Water Stat* database (Mekonnen and Hoekstra, 2010a) in order to re-introduce volumes of grey water which remain of interest in this context.

Table 5.1. The top 15 tea producing countries 2013

Countries	Production quantity 2013 (tonnes) [a]	% contribution to global production [a]	Yield (tonne/ha) [a]	Country average water footprint (m^3/tonne) [b]		
				Green	Blue	Grey
China	1,924,457	36.00%	1.21	9,277	798	1,496
India	1,208,780	22.61%	2.36	4,778	1,332	360
Kenya	432,400	8.09%	2.40	4,061	4	89
Sri Lanka	340,230	6.36%	1.69	10,306	-	1,421
Viet Nam	214,300	4.01%	1.94	12,490	191	485
Turkey	212,400	3.97%	3.06	2,296	735	160
Iran	160,000	2.99%	7.20	1,827	8,791	444
Indonesia	148,100	2.77%	1.33	11,172	-	257
Argentina	105,000	1.96%	3.05	7,641	1,222	246
Japan	84,800	1.59%	2.06	4,996	55	2,081
Thailand	75,000	1.40%	3.85	36,622	5,836	1,774
Bangladesh	64,000	1.20%	1.21	-	-	-
Malawi	54,000	1.01%	2.33	4,642	3,968	-
Uganda	53,000	0.99%	2.09	5,842	-	2
Burundi	41,817	0.78%	5.05	10,816	-	2
Others	227,239	4.25%	-	-	-	-
World	5,345,523	100%	-	7,322	898	726

Source: [a] FAOSTAT (2016). [b] Mekonnen and Hoekstra (2010a). Note: these figures are for the purpose of broad country comparison and have not been used in the specific analysis in this chapter.

5.2 Supply chain water footprint directly associated with inputs

The primary ingredient in the production of a 50 gram box of tea is processed black tea leaves, the raw material and process water footprints associated with which, are detailed in Table 5.2 below for each of the four countries at stage 1. As referred to above, the data from Jeffries *et al.* (2012) on the raw material footprint of tea has been substituted in Table 5.2 for data from Mekonnen and Hoekstra (2010a). This has been done to include the volumes of grey water that correspond to the tea crop.

Table 5.2. Water footprint of black tea

	Water footprint m^3/tonne of raw material [a]				Process water requirement m^3/tonne [b]			
	Green	Blue	Grey	**Total**	Green	Blue	Grey	**Total**
Kenya (Kericho)	4,117	5	94	**4,216**	0	0.12	0	**0.12**
Kenya (Nyeri)	3,721	4	72	**3,797**	0	0.12	0	**0.12**
Indonesia (Agrabinta)	11,354	0	277	**11,631**	0	0.12	0	**0.12**
India (Kotagiri & Coonor)	4,863	1,632	298	**6,793**	0	0.12	0	**0.12**

Source: [a] Mekonnen and Hoekstra, 2010a. [b] Estimate derived from process water requirement and product fraction listed in Jeffries *et al.* (2012). Note, as mentioned above, grey water was excluded by Jeffries *et al.* (2012) and as a result is not included in the process water requirement here.

Whilst Jeffries *et al.* (2012) do not explicitly record the percentage that tea, from each of the four countries of origin at stage 1, constitutes of the end-product (i.e. the blend of tea in the end product), it is possible to extrapolate this information as shown in Table 5.3 below. This accords with the limited information that Jeffries *et al.* (2012) do refer to regarding the tea blend as they mention that tea from India represents approximately 10%.

Table 5.3. Composition of tea in the end-product

Kenya (Kericho)	Kenya (Nyeri)	Indonesia (Agrabinta)	India (Kotagiri & Coonor)
67%	7%	17%	10%

Source: Extrapolated from Jeffries *et al.* (2012).

In addition to tea, Jeffries *et al.* (2012) also estimate the water footprint associated with packaging inputs (tea bag materials and other packaging). For one box of tea, the associated water footprint was estimated at 29.6 litres, the vast majority of which is green water. As referred to in Chapter Three, given that Jeffries *et al.* (2012) were not able to define a specific location for the generic inputs that comprise packaging (i.e. the water footprint associated with these inputs forms part of what was referred to earlier as the non-geographically specific footprint), it is assumed here that the associated water burden falls at the packing factory in Belgium.

5.3 Operational water footprint directly associated with inputs

Data for the operational water footprint (0.005 litres/50g tea) has been sourced from Jeffries *et al.* (2012). However, Jeffries *et al.* (2012) did not report how this water footprint component breaks down between the two factory stages i.e. stage 2 and 3. Therefore, it has been assumed here that this component is split evenly between the two factory locations (i.e. Manchester and Brussels).

5.4 Supply chain and operational overhead water footprints

As above, data for the supply chain (1.6 litres/50g tea) and operational (0.003 litres/50g tea) overhead water footprints has been sourced from Jeffries *et al.* (2012). However, again Jeffries *et al.* (2012) were not specific about how these footprint components break down between stages 2 and 3. Therefore, it has again been assumed that these components split evenly between the two factory locations (i.e. Manchester and Brussels).

It should be noted here that the supply chain overhead water footprint, like the water footprint associated with packaging inputs, forms part of the non-geographically specific footprint given that it is comprised of generic items bought and sold on world markets.[33] Consequently, as mentioned in Chapter Three, in this context it is assumed that the water use associated with the supply chain overhead footprint occurs in the factory locations at stages 2 and 3.

5.5 The water footprint of tea consumption

Jeffries *et al.* (2012) estimate that the water footprint linked to the consumption of tea is approximately 5 litres per 50g box, all of which is blue water. This volume is comprised of 2.2 litres of water associated with tea consumption, and 2.8 litres associated with the electricity used to boil the water.[34]

5.6 Out of scope and caveats

As mentioned in section 5.1, because grey water was excluded in the Jeffries *et al.* (2012) study, visibility over degradative water volumes is consequently limited here. However, by sourcing data from Mekonnen and Hoekstra (2010a) on the water use during crop cultivation at stage 1, this has been rectified for the stage in the supply chain that accounts for the greatest use of water resources (approximately 90% of total green and blue water is associated with stage 1). In addition, whilst both the operational and operational overhead water footprint data associated with stages 2 and 3 excludes grey water volumes, given the advanced nature of the countries in which any grey water would occur (i.e. the UK and Belgium), it seems reasonable to assume that any waste water would be returned, via the sewerage network, to a treatment plant and thus that grey water would be zero. Furthermore, the tea packing and blending processes at stage 2 and 3, with which the operational and operational overhead footprints are associated, both consume negligible volumes of water and the packing and blending of tea are not processes which give rise to water borne pollutants. Conversely, the water footprint associated with packaging inputs, and the supply chain overhead footprint, may have an

[33] Jeffries *et al.* (2012) accounted for the building materials (concrete and steel), paper and energy used in the factories at stage 2 and 3 of the supply chain.
[34] The water use allocated to tea consumption assumes that of 35% of ingested water evaporates through breathing and perspiration. The remaining water is assumed to be returned to the same basin that it was extracted from thus constituting a non-consumptive use (Jeffries *et al.*, 2012). Based on a typical 250g box of tea containing 80 bags which has been consulted here for reference, a 50g box would contain 16 bags and therefore account for approximately 137.5 ml per bag (i.e. 2,200 ml/16 bags).

associated grey water footprint. Given their small size in volume terms though, lack of visibility on the grey water associated with these components is a recognised limitation in this context.

5.7 Total water footprint

Table 5.4 and 5.5 below set out the total water footprint for 50g (320 litres) and one tonne (6,400 m^3) of tea respectively.

Table 5.4. Water footprint of one box containing 50g of tea (litres)

Supply chain stage	Location	Description	Water footprint component	Green	Blue	Grey	**Total**	% of total
1 [a]	Kenya (Kericho)	Tea cultivation and processing	Supply chain	137.17	0.16	3.13	**140.47**	43.9
1 [a]	Kenya (Nyeri)	Tea cultivation and processing	Supply chain	12.4	0.01	0.24	**12.65**	3.9
1 [a]	Indonesia (Agrabinta)	Tea cultivation and processing	Supply chain	94.58	0	2.31	**96.89**	30.3
1 [a]	India (Kotagiri & Coonor)	Tea cultivation and processing	Supply chain	24.4	8.19	1.5	**34.08**	10.6
2 [b]	UK (Manchester)	Blending	Supply chain overhead	0.45	0.35	0	**0.8**	0.25
2 [b]	UK (Manchester)	Blending	Operational	0	0.0025	0	**0.0025**	>0.1
2 [b]	UK (Manchester)	Blending	Operational overhead	0	0.0015	0	**0.0015**	>0.1
3 [c]	Belgium (Brussels)	Packaging	Supply chain	29	0.6	0	**29.6**	9.2
3 [b]	Belgium (Brussels)	Blending	Supply chain overhead	0.45	0.35	0	**0.8**	0.25
3 [b]	Belgium (Brussels)	Blending	Operational	0	0.0025	0	**0.0025**	>0.1
3 [b]	Belgium (Brussels)	Blending	Operational overhead	0	0.0015	0	**0.0015**	>0.1
4 [d]	Belgium (Brussels)	Tea consumption	N/A	0	5	0	**5**	1.6
Total				**298.45**	**14.67**	**7.17**	**320.30**	**100**

Source: [a] Mekonnen and Hoekstra (2010a). [b] Jeffries *et al.* (2012). Note: as referred to above, this assumes that the supply chain overhead, operational and operational overhead water footprints are split evenly between the production facilities in Manchester (stage 2) and Brussels (stage 3). [c] Jeffries *et al.* (2012). Note: as referred to above, this assumes that the water burden associated with packaging inputs is located in Brussels. [d] Jeffries *et al.* (2012). Note: as referred to above, this assumes that tea consumption occurs in Brussels.

Table 5.5. Water footprint of one tonne of tea (20,000 boxes) (m^3/tonne)

Supply chain stage	Location	Description	Water footprint component	Green	Blue	Grey	**Total**	% of total
1 [a]	Kenya (Kericho)	Tea cultivation and processing	Supply chain	2,743.5	3.28	62.54	**2,809.32**	43.9
1 [a]	Kenya (Nyeri)	Tea cultivation and processing	Supply chain	247.95	0.24	4.79	**252.98**	3.9
1 [a]	Indonesia (Agrabinta)	Tea cultivation and processing	Supply chain	1,891.64	0.02	46.16	**1,937.82**	30.3
1 [a]	India (Kotagiri & Coonor)	Tea cultivation and processing	Supply chain	488	163.74	29.94	**681.68**	10.6
2 [b]	UK (Manchester)	Blending	Supply chain overhead	9	7	0	**16**	0.25
2 [b]	UK (Manchester)	Blending	Operational	0	0.05	0	**0.05**	>0.1
2 [b]	UK (Manchester)	Blending	Operational overhead	0	0.03	0	**0.03**	>0.1
3 [c]	Belgium (Brussels)	Packaging	Supply chain	580	12	0	**592**	9.2
3 [b]	Belgium (Brussels)	Blending	Supply chain overhead	9	7	0	**16**	0.25
3 [b]	Belgium (Brussels)	Blending	Operational	0	0.05	0	**0.05**	>0.1
3 [b]	Belgium (Brussels)	Blending	Operational overhead	0	0.03	0	**0.03**	>0.1
4 [d]	Belgium (Brussels)	Tea consumption	N/A	0	100	0	**100**	1.6
Total				**5,969.09**	**293.44**	**143.43**	**6,405.96**	**100**

Source: [a] Mekonnen and Hoekstra (2010a). [b] Jeffries *et al.* (2012). Note: as referred to above, this assumes that the supply chain overhead, operational and operational overhead water footprints are split evenly between the production facilities in Manchester (stage 2) and Brussels (stage 3). [c] Jeffries *et al.* (2012). Note: as referred to above, this assumes that the water burden associated with packaging inputs is located in Brussels. [d] Jeffries *et al.* (2012). Note: as referred to above, this assumes that tea consumption occurs in Brussels.

As shown in Tables 5.4 and 5.5, nearly 90% of the water footprint of tea is attributable to the tea crop at stage 1. Indeed, in absolute terms based on *total* volume data, tea cultivation in Kericho (43.9%) and Agrabinta (30.3%) appear to be the areas of greatest water impact. Alternatively, based on the consumption of limited global blue water resources, Kotagiri and Coonor appear to be of most concern, representing 98% of blue water consumption at stage 1, and 56% of total blue consumption across stages 1 to 4. However, as we have seen in the previous chapter, water footprinting also takes into account the vulnerability of local water systems using the water stress index in order to inform scenarios such as these. The water stress index measures the ratio of total annual water withdrawals in an area to total annual water availability and it can be used to assess the impact of blue water usage in the supply chain and thus identify ‘hotspots.’ Following the approach set out in Jeffries *et al.* (2012, p.159), a hotspot occurs where ‘the blue water footprint of products is large and where water scarcity is high,’ the latter being defined as where it exceeds a value 0.6. Table 5.6 below sets out the water stress values for each of the sourcing locations at stage 1 using data from the World Resources Institute. Table 5.6 suggests that Kotagiri and Coonor in India is a potential hotspot given the fact it supplies 10% of the tea at stage 1 and exhibits a water stress value of 0.66. This accords with the analysis set out by Jeffries *et al.* (2012) who also identified Kotagiri and Coonor, as a potential hotspot, albeit using alternative water stress data. However, choices such as these regarding which geographic area exhibits the greatest concern may be further aided by a focus on the monetary valuation of these water volumes, a subject to which Part B now turns.

Table 5.6. Baseline water stress values for stage 1 tea sourcing regions

Country	State/region	Baseline water stress	% of tea sourced from
Kenya – Kericho	Kericho	0.04	67
Kenya – Nyeri	Nyeri	0.12	7
Indonesia	Agrabinta	0.09	17
India	Kotagiri & Coonor	0.66	10

Source: World Resources Institute (2013).

Part B – Unit water values along the supply chain

Having looked at the volumes of water that are consumed and degraded along the supply chain in the production of tea, Part B now turns to the monetary value of this water and what consideration of this can add to water footprint assessment.

Part B is structured as follows: section 5.8 estimates the value of blue water used in tea production; section 5.9 estimates the value of grey water, and section 5.10 comments on the suitability, in this context, of the approach to estimating green water values that was set out in Chapter Three (Part Three). As mentioned in Chapter Three, the focus here will be the direct use value that accrues to these volumes of water when they are extracted from the stream and used in agricultural, industrial and municipal settings. Section 5.11 draws together the preceding sections and looks at the implications of the analysis. This will include a number of sensitivities which have been conducted on the values that are presented here, in part to reflect the potential of in-stream values to alter the inferences that are arrived at. Finally, section 5.12 concludes the chapter and summarises the analysis in Parts A and B.

5.8 Blue water value

The direct use values attributed to blue water at each of the four stages of the tea supply chain will be considered in reverse order below, starting with stage 4 and the blue water used during tea consumption.

Consumer use phase (stage 4)

As mentioned in section 5.5, the water used in the consumer use phase is split between tea consumption (44%) and the water associated with the electricity that is needed to boil the kettle (56%). As set out in Part Three of Chapter Three, a standard two-part formula for a simple household demand function has been utilised to value the water used in the home to consume tea (i.e. the 44%). The first part of the formula derives the value of treated water delivered to the home; the second part estimates the net consumer surplus which is equivalent to the value of raw water in the stream. The two parts of the formula are repeated directly below. In conjunction with the inputs in Table 5.7, an at site value of \$8.20 (part 1) and an at source value of \$0.67 (part 2), both per cubic metre, were estimated.

Part 1

$V = [(P\ x\ Q_1^{\frac{1}{E}})\ /\ (1 - \frac{1}{E})]\ *\ [(Q_1^{1-\frac{1}{E}})\ - (Q_2^{1-\frac{1}{e}})]$ Young and Loomis (2014)

Part 2

$CS = V - [(P_1)(Q_1 - Q_2)]$

Where:

E = Elasticity

P = Price

Q = Quantity

Table 5.7. Residential water value – Demand function inputs

Input	Value	Source
Q1	96.3 litres per person per day (10% reduction on Q2).	
Q2	107 litres per person per day; 39 m^3 per annum.	Environment Agency (2008)
Price (2014 USD)	7.53 (rate for highest use block 30+ m^3 in Brussels)	Global Water Intelligence (2016)
Price elasticity estimate	-0.62	Vanhille (2012)
At site value (2014 USD per m^3)	8.20	
At source value (2014 USD per m^3)	0.67	

Industrial water use (stage 2 and 3)

The water used by industry in Manchester and Brussels, in the direct operations of each factory (i.e. not the *operational overhead* or the *supply chain overhead* water footprints), has been valued with reference to the two sources highlighted in Part Three of Chapter Three. There it was argued that Wang and Lall (2002) and Bruneau (2007) provide the most robust and appropriate estimates of the value of water *consumed* in a wide variety of industries. Table 5.8 below shows the values that Wang and Lall (2002) and Bruneau (2007) have derived specifically for water that is consumed by the food industry. In what follows, the average of the two values shown in Table 5.8 ($2.39) will be utilised. As mentioned in Chapter Three (Part Three), No value will be assigned to the *operational overhead* and *supply chain overhead* water footprints here. This is because, in the case of the latter, it is made up of a *variety* of goods used in the Manchester and Brussels factories which cannot be directly associated with one final product. Given this variety (including as mentioned earlier, building materials, paper and energy), there is not sufficient detail to select an appropriate value to be transferred. Similarly, the operational overhead water footprint may be used for a variety of purposes within each factory and so is likewise excluded from the following value calculations. However, it must be remembered that both footprint components represent less than 1% of the total water footprint, and as mentioned elsewhere, given that they are not geographically specific, they will never be a relevant change variable when comparing water values in different regions.

Table 5.8. Food industry values used in the tea supply chain case study

Supply chain location at Stage 2 (Policy site)	Source	Method	Value type	Water volume measure	Original value m^3 (currency)	2014 USD per m^3
UK and Belgium	Wang & Lall (2002)	Production function	MV	Consumption	2.57 (Yuan)	1.87
UK and Belgium	Bruneau (2007)	Alternative cost	AV	Consumption	2.5 (CAD)	2.92
						2.39 (Average)

MV = Marginal Value. AV = Average Value.

Agricultural water use (stage 1)

Table 5.9 below sets out the values which have been selected from the literature for each of the three locations at stage 1. In the case of Kenya, the values selected are for irrigation in the Kerio Basin which is proximate to both Nyeri and Kericho. Likewise, the values selected for Indonesia are for agricultural water use in East Java which is contiguous to, and on the same island as, West Java where Agrabinta is located. However, the only values available in the literature for India are for the northern and eastern parts of the country, whereas Kotagiri and Coonor are both located in the south. As a result, in the absence of better data for India, the values in Table 5.9 will be utilised but this lack of correspondence between the characteristics of the study and policy sites should be considered a limitation in this context. All of the values in Table 5.9 are for water application or diversion as none were available, for the policy sites, which reflected water consumption. As such, they represent a lower bound estimate of the value of water consumed at each location.

Unlike stages 2 to 4 in the supply chain, which each have a single location, there are three locations for stage 1. Consequently, the *relative* value between stage 1 locations becomes important if the analysis is to compare the impacts of water use at each stage 1 location. As a result, the values presented in Table 5.9 below have been selected because, as far as possible, they are comparing a common scenario. For instance, each of the values in Table 5.9 is for low valued crops (small grains), they are short run and at site, and they have been estimated using techniques which yield an average value. However, whilst every care has been taken to ensure a consistent comparison, Table 5.9 shows that two of the estimates measure diversion as opposed to application, and more broadly,

Table 5.9. Agricultural values used in the tea supply chain case study

Supply chain location at Stage 1 (Policy site)	Source	Method	Value type	At site/ at source	Long run/short run	Water volume measure	Crop value	Original value m^3 (currency)	2014 USD per m^3	Study location (Study site)
Kenya (Kericho and Nyeri)	Kiprop *et al.* (2015)	Farm crop budget	AV	At site	Short	Application	Low (millet)	4.3 (Kenyan Shilling)	0.11	Kenya (Kerio Basin)
Kenya (Kericho and Nyeri)	Kiprop *et al.* (2015)	Farm crop budget	AV	At site	Short	Application	Low (sorghum)	11.28 (Kenyan Shilling)	0.30	Kenya (Kerio Basin)
Kenya (Kericho and Nyeri)	Kiprop *et al.* (2015)	Farm crop budget	AV	At site	Short	Application	Low (maize)	14.87 (Kenyan Shilling)	0.40	Kenya (Kerio Basin)
AVERAGE									**0.27**	
Indonesia (Agrabinta)	Hellegers & Perry (2004)	Farm crop budget	AV	At site	Short	Application	Low (multiple – unclear)	0.04 (USD)	0.05	Indonesia (Brantas Basin - East Java)
Indonesia (Agrabinta)	Rodgers & Hellegers (2005)	Farm crop budget	AV	At site	Unclear	Application	Low (rice)	0.02 – 0.05 (USD)	0.03 – 0.07	Indonesia (Brantas Basin - East Java)
Indonesia (Agrabinta)	Rodgers & Hellegers (2005)	Farm crop budget	AV	At site	Unclear	Application	Low (maize)	0.08 – 0.11 (USD)	0.11 – 0.15	Indonesia (Brantas Basin - East Java)
AVERAGE									**0.08**	
India (Kotagiri & Coonor)	Rodgers *et al.* (1998)	Yield Comparison	AV	At site	Short	Diversion	Low (rice and wheat)	0.019 (USD)	0.03	Northern India (Haryana)
India (Kotagiri & Coonor)	Rodgers *et al.* (1998)	Yield Comparison	AV	At site	Short	Diversion	Low (unclear)	0.027 (USD)	0.04	Eastern India (Jamshedpur)
India (Kotagiri & Coonor)	Hellegers & Perry (2004)	Farm crop budget	AV	At site	Short	Application	Low (multiple – unclear)	0.04 (USD)	0.05	Northern India (Haryana)
AVERAGE									**0.04**	

AV = Average Value. Values converted from local currency to 2014 USD using World Bank PPP exchange rates and Implicit Price Deflator (Appendix 3). See Chapter Three.

each of the estimates is sensitive to the exact crop and, for example, the exact cost components used in the farm crop budget, many of which are not fully discernible in the respective sources. As such, again, the values in Table 5.9 should be considered indicative only; they would need to be investigated using fully consistent primary valuation techniques in each location if a policy relevant action was contingent on them. Finally, whilst tea is not a low valued crop, values for higher valued crops, in each location, were not available and therefore, again, the values in Table 5.9 represent a lower bound value in this context.

Figure 5.2 below sets out the unit values that have been assigned to blue water consumption at each stage along the tea supply chain, and the value of the specific volume of blue water used at each stage (one tonne scenario). In this scenario, an average of the three low valued crops in Kenya has been used. Likewise, in Indonesia and India, an average of the values recorded has been selected. However, section 5.12 will undertake a number of sensivity analyses to reflect the range of values on display in Table 5.9, and more broadly, what is an unknown level of transfer error at each stage 1 location.

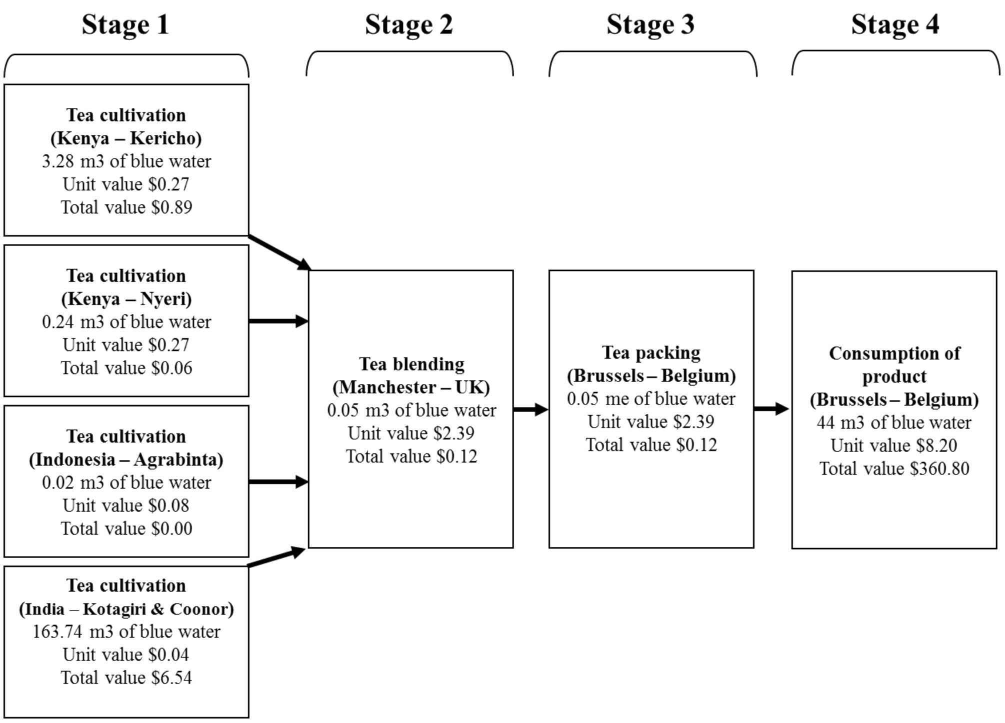

Figure 5.2. Blue water values assigned to each stage of the tea supply chain (one tonne scenario). Note: 1) values for stages 2 and 3 refer to the operational water footprint only, and 2) value for stage 4 refers to the 2.2 litres of water associated with tea consumption in the home.

As shown in Table 5.10 below, 98% of the total at site value of blue water consumed in the supply chain occurs during the consumer use phase (stage 4) even though this only accounts for approximately 21% of the volume of blue water. This disparity is primarily driven by the comparatively high unit value assigned to municipal use at stage 4 and ensures that whilst 77% of the total volume of blue water use occurs in India, this only accounts for 2% of the total value. Looking at stage 1 in isolation, Kericho accounts for 2% of the volume of irrgation water used, but this represents 12% of total value given what is, by comparison to Indonesia and India, the relatively high unit value assigned to irrigation water in Kenya. Similarly, irrigation water used in India accounts for 98% by volume but only 87% by value given the low unit value that prevails in India. The total direct use value of blue water consumed in the production of one tonne of tea is $369, or, using the nominal exchange rate in mid 2017 (1 USD = 0.77 GBP), approximately £284.

Table 5.10. Blue water value and volume distribution in the tea supply chain (one tonne scenario)

Stage (location)	Volume of blue water (m^3)	Unit value (USD 2014)	Value of blue water consumed (USD 2014)	% of total blue water volume	% of total blue water value	% of stage 1 volume	% of stage 1 value
1 (Kenya – Kericho)	3.28	0.27	0.89	2	<1	2	12
1 (Kenya – Nyeri)	0.24	0.27	0.06	<1	<1	<1	1
1 (Indonesia)	0.02	0.08	0.00	<1	<1	<1	<1
1 (India)	163.74	0.04	6.54	77	2	98	87
2 (UK – Manchester)	0.05	2.39	0.12	<1	<1		
3 (Belgium – Brussels)	0.05	2.39	0.12	<1	<1		
4 (Belgium – Brussels)*	44	8.20	360.80	21	98		
Total	**211.38**		**368.53**	**100**	**100**	**100**	**100**

5.9 Grey water

As referred to previously in Chapters Three and Four, it is assumed here that the unit value of grey water degradation is equal to the unit value of blue water consumption. This assumption has been made because grey water refers to the volume of blue water that is necessary to assimilate or abate pollution. As we have seen, blue water consumption impacts a variety of in-stream ESS (waste assimilation, wildlife habitat and recreation) and off-stream extractive uses. However, only the values associated with of off-stream extractive uses are available here so the unit values of grey water are identical to those presented in the previous section. Figure 5.3 below re-states the applicable unit values and sets out the value of grey water along the supply chain based on these unit value estimates. As mentioned in section 5.6, there is no grey water consumed in stages 2 to 4 of the supply chain.

It should be noted here that the assumptions regarding grey water that were discussed in Chapter Three (Part Three), principally that to treat it as a real as opposed to theoretical volume of water we need to assume that there is not more pollution than assimilative capacity in the receiving water body, are called into question here. Indeed, the work of Liu *et al.* (2012) suggests that, broadly, excessive nitrogen and phosphorous discharges are more prevalent in the southern hemisphere, and that high general water pollution levels are to be found in tropical-subtropical areas. This obviously suggests that all three countries at stage 1 may potentially not have sufficient assimilative capacity. However, in the absence of more specific data, and given the level of spatiotemporal detail that the method here is adhering to, this is a recognised limitation in this context and one which would need to be addressed using primary valuation techniques should decision relevant values be required.

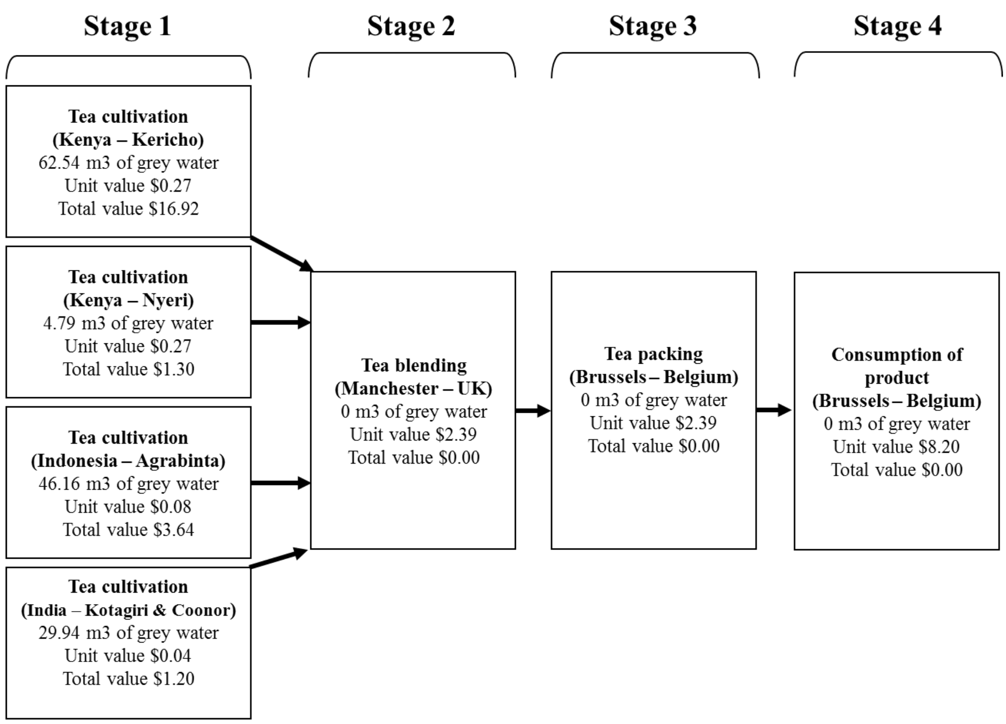

Figure 5.3. Grey water values assigned to each stage of the tea supply chain (one tonne scenario).

Table 5.11 below presents the total value of the grey water in the tea supply chain. Owing to the disparities in unit values between locations that were noted above, grey water in Kericho represents 73% of total value but only 44% by volume. Similarly, grey water in India represents 21% by volume but only 5% of total value. The total value of the grey water associated with one tonne of tea is $23, or, using the nominal exchange rate mentioned above, £18.

Table 5.11. Grey water value and volume distribution in the supply chain (one tonne scenario)

Stage (location)	Volume of grey water (m^3)	Unit value (USD 2014)	Value of grey water degraded (USD 2014)	% of total grey water volume	% of total grey water value
1 (Kenya – Kericho)	62.54	0.27	16.92	44	73
1 (Kenya – Nyeri)	4.79	0.27	1.30	3	6
1 (Indonesia)	46.16	0.08	3.64	32	16
1 (India)	29.94	0.04	1.20	21	5
Total	**143.43**		**23.05**	**100**	**100**

5.10 Green water

Part Three of Chapter Three set out the approach to valuing green water in light of the available valuation data collected during this study. By way of a recap, green water in this context is not rain water as such but the water that is evapotranspired by the potato crop during its growth phases, or, in other words, it is the volume of water that is usefully absorbed by the crop. As such, it had been anticipated that values for irrigation water *consumed* by the crop would be used as a proxy for the value of green water. However, these were not available in the supply chain locations in stage 1, and as a result, the value of green water will be assumed to be equivalent to the *at source* value of artificially applied irrigation water. In order to estimate at source values, the difference between the mean and median at site and at source values for irrigation water in the USA and ROW value databases presented in Chapter Three (Part Two), as a whole, were assessed. The largest difference (USA database; mean value) showed that at source values were typically 60% of at site values; the smallest difference (ROW database; median value) showed that at source values were typically 80% of at site values. As a result, these two measures were used to deflate the at site blue water values used above to provide an estimate of the at source value at each stage 1 location. Sensitivity 1 below (or S1) reflects the at source value at 60% of the at site value; sensitivity 2 (or S2) reflects 80%.

In many ways this is a crude estimate of the value of green water. However, as mentioned earlier, Aldaya *et al.* (2010a) point to the contemporary significance of green water in the international trade in crops, and thus ensuring that the value of green water is incorporated here in some way if possible, is important. What is more, by using a measure of the at source value of water diverted or applied, this is in many ways a conservative estimate of the value of water that is consumed, and thus becomes more defensible.

Figure 5.4 below sets out the unit values of green water and the total value of green water consumed at each stage of the supply chain. There is no green water consumed in stages 2 to 4 of the supply chain.

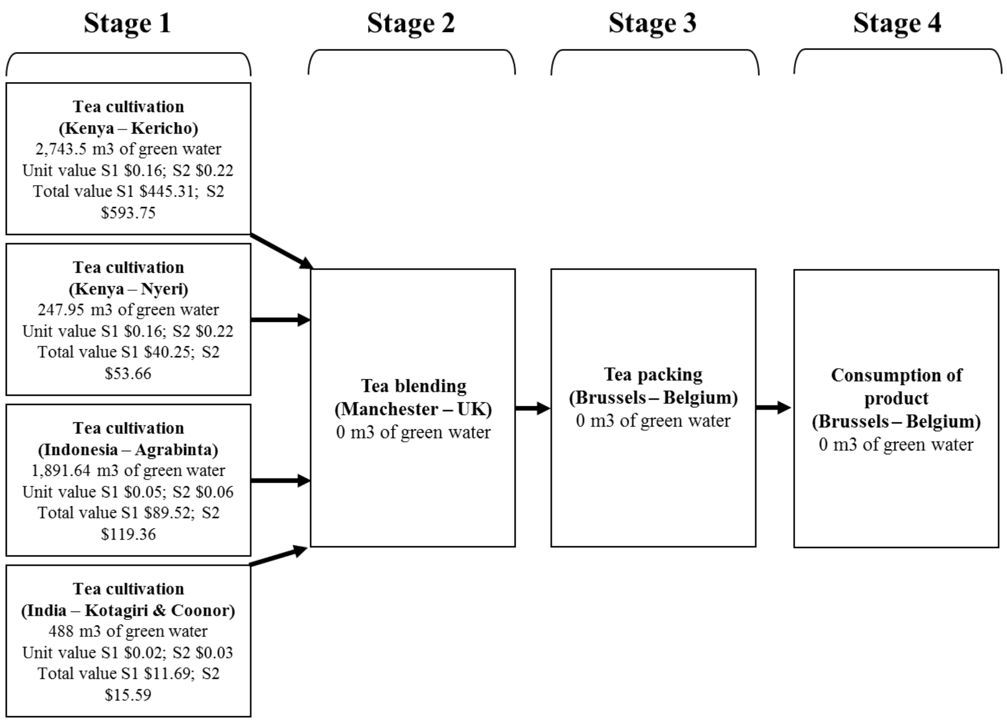

Figure 5.4. Green water values calculated for each stage of the tea supply chain (one tonne scenario).

Table 5.12 below presents the total value of green water in the tea supply chain. As discussed in the previous and forthcoming chapters on the pasta and potato crisps case studies, there is a clear issue associated with including the value of green water at the level suggested by the method used here. This issue is slightly less clear cut in this context because the water footprint volume data from Mekonnen and Hoekstra (2010a) that forms the basis of this case study (Table 5.2) is applicable to tea which has been processed from raw tea leaves. In addition, the value of raw tea leaves prior to processing into tea, is not readily available. Therefore, it is not possible to compare the price of a tonne of raw tea leaves with the estimated value of green water consumed in the production of one tonne of raw tea leaves. Nonetheless, given that one tonne of processed tea, between 2013 and 2015, had an average price per tonne of approximately $2,700, the idea that the value of green water might represent between 22% (S1 $587) and 29% (S2 $782) of this amount seems unlikely (World Bank, 2017). Indeed, a farmer would be unlikely to be willing to pay for green water at these levels, on top of the value of blue water for which we only have a lower bound estimate, and as indicated elsewhere, ultimately the value of water in agriculture is a derived demand and driven by the crop price. Therefore, given the analysis here, but in particular, the analysis presented in the other two case studies, the value of green water has been excluded and the approach to valuing green water will be revisited in Chapter Seven when the conclusions and recommendations from the project as a whole are discussed.

Table 5.12. Green water value and volume distribution in the supply chain (one tonne scenario)

Stage (location)	Volume of green water (m^3)	Unit value S1 (USD 2014)	Unit value S2 (USD 2014)	Value of green water consumed S1 (USD 2014)	Value of green water consumed S2 (USD 2014)	% of total green water volume	% of total green water value
1 (Kenya – Kericho)	2,743.5	0.16	0.22	445.31	593.75	51	76
1 (Kenya – Nyeri)	247.95	0.16	0.22	40.25	53.66	5	7
1 (Indonesia)	1,891.64	0.05	0.06	89.52	119.36	35	15
1 (India)	488	0.02	0.03	11.69	15.59	9	2
Total	**5,371.09**			**586.77**	**782.36**	**100**	**100**

5.11 Implications

Based on the analysis above, Table 5.13 below sets out the total value associated with the water footprint of one tonne of tea as an end product. This is based on approximately 90% of the total water consumed in the supply chain as the remaining 10%, referring to the *operational* and *supply chain overhead* water footprints, and the water associated with packaging and electricity production, encompass too much variation to meaningfully assign a value to them. As shown the, the value of the blue and grey water consumed in the production of one tonne of tea is $392.

Table 5.13. Total value of the blue and grey water used to produce one tonne of tea (finished goods)

Water footprint component	Value/cost USD 2014	Value/cost GBP
Blue	369	284
Grey	23	18
Total value	392	302

Table 5.14 below shows how total value breaks down by supply chain stage thus directly addressing RQ2. As presented, it is the high at site unit value assigned to stage 4, combined with the substantial volumes of blue water (44 m^3 per tonne) that are used in the consumption of tea, that ensure that this stage accounts for over 90% of total water value in the supply chain. Indeed, the value of water at stage 4 obscures the differences in value between the locations at stage 1, imbalances in which, highlight the real merit of a monetary valuation approach such as this and the trade-offs that it enables.

Table 5.14. Total blue and grey water value by supply chain stage

Stage (location)	% of total blue and grey water value
1 (Kenya – Kericho) Blue water	<1
1 (Kenya – Kericho) Grey water	4
1 (Kenya – Nyeri) Blue water	<1
1 (Kenya – Nyeri) Grey water	<1
1 (Indonesia) Blue water	<1
1 (Indonesia) Grey water	1
1 (India) Blue water	2
1 (India) Grey water	<1
2 (UK) Blue water	<1
3 (Belgium) Blue water	<1
4 (Belgium)Blue water	92
Total	**100**

In order to rectify this, and address the fact that different quantities of tea from each of the four stage 1 locations end up in the final tea blend (see Table 5.3), Table 5.15 sets out the value of the blue and grey water associated with a common quantity (one tonne) of tea cultivated in each location (i.e. not based on the tea blend in Table 5.3 and not including the water associated with stages 2 to 4). This is based on the water volumes noted in Table 5.2 previously and the blue and grey water unit values assigned to stage 1 that have been referred to in sections 5.8 and 5.9.

Table 5.15. Total value of the blue and grey water used to produce one tonne of tea in each location

Stage 1 location	Blue water (m^3)	Grey water (m^3)	Unit value (USD 2014)	Total value of blue water (USD 2014)	Total value of grey water (USD 2014)	Total value of blue and grey water (USD 2014)
India	1,632	298	0.04	65.16	11.90	**77.06**
Kenya - Kericho	5	94	0.27	1.35	25.43	**26.78**
Indonesia	0	277	0.08	0	21.85	**21.85**
Kenya - Nyeri	4	72	0.27	1.08	19.48	**20.56**

Given that value and WTP reflect the intensity of individuals' preferences for water, unlike inter-sectoral water allocation where the same unit of water should, according to economic theory, be used by the highest valued user, here the optimal sourcing location at stage 1 would exhibit the lowest water value. Alternatively, given that values are no longer in evidence when the water is consumed or degraded, they effectively represent costs, and therefore sourcing from the location with the lowest value would represent the optimal solution. In light of this, it is clear from Table 5.15 that whilst Kenya exhibits the highest unit value of blue and grey water ($0.27 per cubic metre), Nyeri accounts for the lowest volume of blue and grey water consumed (76 m^3) and thus the lowest overall value ($20.56) in *volume adjusted* terms. Nyeri therefore appears to be the optimal sourcing location, followed by Indonesia, Kericho, and last of all, India. This accords with the volumetric perspective in the sense that Nyeri pollutes and consumes the lowest total volumes of water and India pollutes and consumes the most. In addition, it is also in agreement with the insights gained from the water scarcity data that identified India as a potential hotspot. However, investigating this further, it is only by including the monetary value of the water volumes concerned that it becomes clear that Indonesia is a more favourable sourcing location when compared Kericho, despite the fact that the former pollutes and consumes nearly three times the volume of water when compared to the latter. In addition, it is only by including monetary values that the cost savings that would be realised if tea was sourced from one location versus another can be identified. For example, this saving would amount to $56.50 if a tonne of tea was sourced from Nyeri as opposed to India.

However, again, these conclusions are based on limited evidence regarding the unit values which prevail in each geography. As a result, the standard convergent validity techniques that would usually be applied here to estimate transfer error in each location are not feasible. Therefore, given the sensitivity of the conclusions to the precise unit values applied in each location, and the importance of the *relative* differences in unit values between locations, we now move on the sensitivity analysis in order to ascertain the degree of certainty around the conclusions drawn thus far.

5.12 Sensitivity analysis

As with the other two case studies in Chapters Four and Six, two sensitivities will be deployed here. The first looks at the lowest *unit value* in evidence – India – and estimates the increases that would be necessary to this unit value in order for it to be comparable with the other value estimates in Indonesia and Kenya. This is particularly appropriate because, as indicated, the unit value for India, whilst the best available, has been taken from regions of the country which differ from the policy site. The second will look at the *volume adjusted values* set out in Table 5.15 above and estimate the increases in value that would be necessary in the lowest valued location – Nyeri (Kenya) – for it to be comparable with the other volume adjusted values. As part of this second sensitivity, the likelihood that in-stream values could account for any increase in values will also be commented on.

Sensitivity 1

Figure 5.5 below sets out the absolute unit value and unit value percentage increases (right hand column) that would be necessary for the lowest unit value in India to be comparable with the unit values in Indonesia and Kenya. For example, there would need to be a 98% increase in the Indian value for it to be comparable with the Indonesian value, or a 578% increase for it to be comparable with the Kenyan value. This is based on the standard formula for estimating transfer error as set out below, except that the observed and transferred values refer to separate locations (Czajkowski and Scasny, 2010):

$$E_{TR} = \frac{WTP_{transferred} - WTP_{observed}}{WTP_{observed}}$$

In addition, Figure 5.5 also presents the unit value and percentage decreases (left hand column) that would be necessary for the unit values in Kenya and Indonesia to be comparable with the unit value in India. For example, there would need to be an 85%

decrease in the Kenya unit value, or a 49% decrease in the Indonesian value, for them to be comparable with the Indian unit value.

% decrease	Unit value decrease	Country	USD 2014	Unit value increase	% increase
-85%	-0.23	Kenya	0.27	0.23	578%
-49%	-0.04	Indonesia	0.08	0.04	98%
		India	0.04		

Figure 5.5. Unit value sensitivities/transfer errors (1).

Czajkowski and Scasny (2010) suggest that the majority of transfer errors are in the 0-200% range. Given this, and the values in Figure 5.5, one clear conclusion is that, from a *unit value perspective*, the optimum sourcing location would likely not be Kenya given the necessary transfer error of 578% compared to India. This conclusion is reinforced when considering Figure 5.6 below which presents the percentage increases or decreases that would be necessary for a unit value to be comparable, not with the Indian value, but with the unit value closest to it. For example, there would need to be 243% increase in the Indonesian value for it to be comparable with the Kenyan value. Given this, it therefore seems reasonable to conclude that Kenya, form a unit value perspective, would not be the optimum sourcing location. Indeed, referring back to the unit values in Table 5.9, whilst there was some overlap between the lower range Kenyan value and the upper range Indonesian value, on a like for like basis growing maize, the value in Kenya was significantly greater than in Indonesia ($0.40 compared to $0.11 – 0.15).

% decrease	Unit value decrease	Country	USD 2014	Unit value increase	% increase
-71%	-0.19	Kenya	0.27	0.19	243%
-49%	-0.04	Indonesia	0.08	0.04	98%
		India	0.04		

Figure 5.6. Unit value sensitivities/transfer errors (2).

Sensitivity 2

Sensitivity 2 looks at how much the 76 m^3 of blue and grey water used in Nyeri (the location with the lowest *volume adjusted value* in Table 5.15) would have to increase by to be comparable with the other three locations analysed here. Table 5.16, which is derived from Table 5.15 above, presents the difference in volume adjusted value between Nyeri and each of the other locations (column two). Dividing this difference by 76 m^3 provides an indication of how much the unit value would need to increase in value by (column 3) to be comparable with each of the other locations. As shown, given the disparity in unit values between Kenya and Indonesia, despite the fact that the latter accounts for 201 m^3 more blue and grey water in the production of a tonne of tea (Table 5.15), it would only require a small (6%) increase in the unit value in Nyeri for the volume adjusted value to be comparable with Indonesia, thus again highlighting the importance of taking into account values as well as volumes. Conversely, however, it would require a 275% increase, or $0.74 per cubic metre, for the volume adjusted value in Nyeri to be comparable with India. Therefore, it seems reasonable to conclude here that India does not represent the optimum sourcing location in *volume adjusted* terms. Beyond this, Nyeri appears to be the optimum sourcing location form a volume adjusted perspective, but this is relatively sensitive to increases in unit values (a 6% increase would bring it in line with Indonesia whilst a 30% increase would bring it in line with Kericho).

Table 5.16. Sensitivity two – unit value increases in Nyeri

Location	Difference in total value of blue and grey when compared to Nyeri (USD 2014)	Increase in unit value of 76 m^3 of blue and grey (USD 2014)	% increase in $0.27 unit value
Indonesia	1.29	0.017	6%
Kenya (Kericho)	6.22	0.082	30%
India	56.50	0.743	275%

In addition, the requisite unit value increases (column 3) can be compared with the instream value scale presented in Chapter Three. Based on the minimum, median and maximum combined waste assimilation, wildlife habitat and recreation values that were evident in the USA (the only country which recorded these values) being present in one location, the instream value scale can be adjusted for relative incomes in Kenya using the formula set out by Czajkowski and Scasny (2010) which assumes an income elasticity of one:

$$WTP_{ps} = WTP_{ss}\left(\frac{\overline{I_{ps}}}{\overline{I_{ss}}}\right)\varepsilon$$

where WTP_{ss} is willingness to pay at the study site, WTP_{ps} is the willingness to pay estimate transferred to the policy site, and I_{ss} and I_{ps} are mean income levels at the study and policy sites. ϵ represents the income elasticity of willingness to pay between the mean income levels at the study and policy sites (Czajkowski and Scasny, 2010). The income data in Table 5.17 below has been used to make this adjustment and the resulting in-stream value scale for Kenya is set out in Table 5.18.

Table 5.17. Relative income levels in Kenya

Country	GNI Per Capita [a]	% of USA GNI Per Capita
USA	52,308.38	100
France	2,157.94	4

[a] Data sourced from UNDP (2014).

Table 5.18. In-stream value scale Kenya (USD 2014 per m3)

Low		Median		High
0.00002	→	0.002	→	0.025

As shown, it is quite clear that the necessary increases in unit values in Nyeri that would be needed in order for the volume adjusted value to comparable with Kericho and India are far beyond equivalent highest in-stream values in the USA (i.e. \$0.082 m^3 and \$0.743 m^3 are both greater than \$0.025 m^3).[35] Given that the in-stream values recorded for the USA are for the most arid parts of the country, and that the USA is, by necessity, at the forefront of unit valuation of water resources in order to improve inter-sectoral water allocation decisions, it seems safe to conclude that the presence of in-stream values in Nyeri is unlikely to produce volume adjusted values which exceed Kericho and India. However, the necessary unit value increase (\$0.017 m^3) for the volume adjusted value in Nyeri to be comparable with Indonesia is within the scope of the in-stream value scale, albeit at the high end. Therefore, it is conceivable that the presence of in-stream values in Nyeri could alter the conclusion that Nyeri is the optimum sourcing location. However, it should be borne in mind that this is not taking into account the possible presence of in-stream values in Indonesia, the presence of which, would further widen the gulf in volume adjusted value between the two locations and thus require the presence of even greater in-stream values in Nyeri

5.13 Conclusion

In conclusion, in Part A the water footprint of a 50g box of tea (320 litres), and 20,000 50g boxes of tea representing one tonne of finished goods (6,400 m^3) was estimated. In addition, it was shown that 90% of this water footprint was associated with the tea crop

[35] As noted in Chapter Three, in-stream ESS values are additional to agricultural values which are net of extraction costs (i.e. the agricultural value is at source). However, given that at source agricultural values were not available here, the in-stream value scale is applied to at site agricultural values on the assumption of minimal/similar extraction costs across stage 1 sourcing locations.

at stage 1. Indeed, in absolute volume terms alone, it was suggested that the cultivation of tea in Kericho (Kenya) and Agrabinta (Indonesia) appear to be the areas of greatest concern, but that Kotagiri and Coonor (India) accounts for the largest share of blue water consumption in the supply chain. However, this analysis was not based on a like for like comparison, but rather the blend of tea that is found in the end product. Part A also suggested that Kotagiri and Coonor represents a potential blue water hotspot based on water stress data which accounted for the availability of water in each of the stage 1 locations.

In Part B, the total value of the blue and grey water used to produce one tonne of finished goods (i.e. 20,000 boxes) was estimated as $392. The vast majority of this value (92%) was associated with the water that is used during tea consumption given the higher unit values linked to municipal water use. However, again this was based on the blend of tea in the end product and therefore was not able to fully illuminate the trade-offs between the multiple stage 1 locations that become apparent when a monetary approach is adopted. Therefore, Part B also undertook a like for like comparison of the value of blue and grey water used to produce a tonne of tea in each location. This showed that whilst Nyeri (Kenya) exhibits the highest blue water *unit value*, in *volume adjusted* terms, because it uses the least blue and grey water, overall it accounts for the least *total* value of water. However, a 6% or 30% increase in the unit value in Nyeri would ensure that the *volume adjusted value* was in line with Indonesia and Kericho (Kenya) respectively (Table 5.16). Given that it would require a 275% increase in the unit value in Nyeri to bring volume adjusted value in line with India, the principal overall conclusion seems to be that India likely represents least optimal sourcing location despite the fact that it has the lowest unit value. This accords with, and in fact is driven by, the volumetric analysis of blue and grey water (where as mentioned India accounts for the largest share of blue and grey water resources) and the analysis of blue water hotspots. However, beneath this overall conclusion, it was only by taking values into account that it becomes apparent, for example, that Indonesia would be a preferred sourcing location when compared to Kericho, despite the fact that the former pollutes and consumes nearly three times the volume of water when compared to the latter. In addition, the sensitivity analysis suggested that the presence of in-stream values in Nyeri was unlikely to influence the conclusion that India is the least optimal sourcing location.

However, all of these conclusions have been reached without the inclusion of green water in the analysis, which it was argued, could not be valued in the way anticipated. In addition, it should be stressed that given the importance of relative differences between unit values at stage 1, they would need to be confirmed using fully consistent primary valuation techniques at each stage 1 location if decision relevant values were required.

Chapters Five and Six, between them, have analysed the pasta and tea supply chains and tested the valuation methodology presented in Chapter Three. However, the following chapter now introduces the potato crisp supply chain case study, volumetric water related data for which, has been sourced from primary sources as detailed in what follows. Indeed, because of this, the potato crisp case study provides a more in-depth description of the precise steps that are followed when accounting for water volumes using the water footprint method.

6. The potato crisp supply chain

This chapter sets out the results of the potato crisp supply chain case study, volumetric water data and supporting information for which, was collected during the first quarter of 2016 directly from the company described below. As mentioned in Chapter Three, this involved discussions with key company personnel and access to internal company documentation.

Part A summarises the potato crisp water footprint i.e. the volumes of green and blue water consumption, and degradative grey water, at each point along the supply chain. Part B summarises the attendant monetary values that have been assigned to these volumes of water.

Part A – The potato crisp water footprint[36]

Part A begins by providing a brief description of the company from which data was sourced for use in this case study (section 6.1). Section 6.2 then defines the production unit that will be the subject of analysis here and provides an overview of the associated supply chain map. Section 6.3 sets out the key foreground processes that occur within each stage of the supply chain. Section 6.4 deploys the simple concept of blue water withdrawal – the traditional measure of company water dependency – and summarises the volumes associated with the primary elements of the potato crisp supply chain. Section 6.5 calculates blue and green water consumption, and degradative grey water burden, for the *supply chain* water footprint directly associated with inputs. Section 6.6 repeats this for the *operational* water footprint directly associated with inputs. Sections 6.7 and 6.8 present the supply chain and operational *overhead* water footprints. Finally, section 6.9 summarises the total water footprint of the potato crisp product.

6.1. Company description

The subject of this case study is a leading manufacturer of crisp products based in the UK ("the company"). Of the company's product lines, 95% of finished goods by weight are accounted for by potato crisps of varying flavours, pack sizes, physical profiles (e.g. flat, wave cut) and sales margins. In addition, the company also produce a number of ancillary crisp products, all of which, except what will be referred to as the *baked*

[36] Note all production data refers to 2015 unless otherwise stated.

ancillary product, utilise raw materials other than potatoes and as a result undergo distinct production processes. The company produces crisps for sale under their own label, as well as on behalf of a number of own-brand clients.

6.2. Product units and supply chain map

This case study is based on two different production quantities of company branded Salted potato crisps (flat profile):

- 150g bag (this product is larger than individual portions of crisps, which in the UK tend to be in 25-35g bag sizes, and consequently is intended for sharing).[37]
- 6,667 150g bags which constitute one tonne of finished goods by weight.[38]

Salted crisps are the most popular flavour of potato crisps offered by the company in terms of sales volume, and 150g bags are the most popular size that this flavour is available in. As a share of total finished goods, Salted 150g bags account for approximately 14% in 2015; other flavours and bag size combinations represent smaller percentages.

Given the relatively small volumes of water consumption associated with similar agri-food products, typically registering in the hundreds of litres per unit, (see for example Ruini *et al.* 2013, Aldaya and Hoekstra, 2010, Chapagain and Orr, 2010, Chapagain and Hoekstra, 2007), larger production quantities are likely to be more meaningful units of analysis when the focus is monetary values which tend to only register in cubic metres (see Part B). As a result, one tonne of finished goods has been included here, which is an unusual unit of analysis in water footprint studies which focus on water volume alone. This will also facilitate an analysis, on a like for like basis, with the tea and pasta water footprint case studies which have also been estimated at the one tonne level.

As shown in Figure 6.1 below, in this scenario potatoes ("primary ingredient") are cultivated (stage 1) either in East Anglia ("Farm 1" or "representative farm") or northern France ("Farm 2").[39] Following this, they are sent for processing (stage 2) and distribution (stage 3), both of which occur in the UK. Farm 1, which produces approximately 5% of the annual quantity of potatoes used in the factory stage, has provided primary data for use in this case study and is regarded by the company as a

[37] 150g refers to the weight of the bag contents i.e. potato crisps.
[38] Refers to the weight of potato crisps only i.e. excludes the weight of packaging.
[39] In reality the company sources potatoes from multiple locations, albeit predominantly within the UK.

substantial supplier. The potatoes that it produces are of the Lady Claire variety – one of six varieties used during stage 2 – which on average accounts for approximately 10% of annual potato inputs during the factory stage. As will be elaborated on in what follows, Lady Claire has been chosen here because it is a specialist crisping variety which is widely used in the industry due to advantageous characteristics such as resistance to bruising, low and reducing sugar levels and the ability to withstand nine months of storage for late season crisp production. Farm 2, located in an area of northern France where the company have historically sourced from, has been included in order to facilitate a comparison of water values across geographical boundaries in Part B. Unlike Farm 1, for Farm 2, secondary data will be relied upon here to estimate the water footprint associated with the production of a generic potato type in this location.

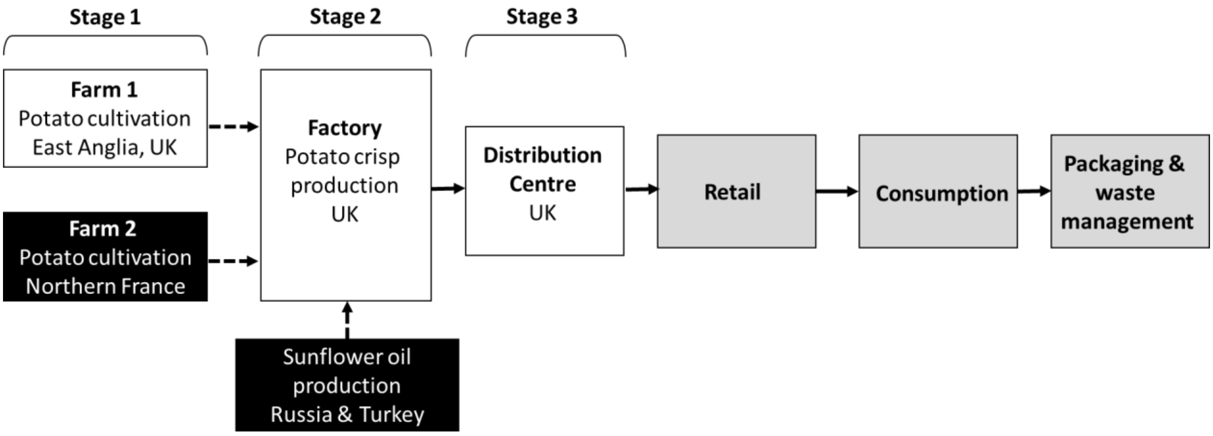

Figure 6.1. Potato crisp supply chain map including principal inputs into production (potatoes and sunflower oil). Stages in grey are excluded from the water footprint analysis. Primary data on water volumes is available for the stages in white. Volumetric data for the stages in black has been obtained from secondary sources as detailed in what follows.

In addition to the volumes of water used in potato cultivation, the volumes of water used in the production of sunflower oil, the other principal ingredient, will also be included in this analysis and thus provide additional points of geographical distribution for investigation in Part B. Whilst the company did not have visibility over the geographical origin of the sunflower oil used, for the purposes of this research, it has been assumed that sunflower oil ultimately originates from either Russia (Krasnodar Krai)[40] or Turkey (Edirne).[41] Whilst both of these countries constitute two of the top five global producers

[40] Krasnodar Krai is a major sunflower seed growing area in Russia located with the Southern Federal District (USDA, No Date).
[41] Edirne is a province within Trakya which is the main sunflower producing region in Turkey (Kaya & Durak, 2007).

(see Table 6.1 below), Turkey has been included here in favour of Ukraine and Argentina because, unlike the other countries in Table 6.1, sunflower oil production in Edirne uses substantial quantities of blue water in cultivation rather than being predominantly a rain fed crop. Given that the primary focus of this research is methodological development, the selection of Edirne enables the greatest degree of variation to be captured in the testing of this methodology.

As with the potatoes grown by Farm 2, the water usage associated with sunflower oil production in both Russia and Turkey will be estimated using secondary sources.

Table 6.1. Top five sunflower oil producing countries 2013

Country	Production (tonnes/year)	% of world production
Russian Federation	3,284,000	26
Ukraine	2,302,801	18
Argentina	1,074,700	9
Turkey	875,445	7
France	578,800	5

Source: FAOSTAT (2016).

6.3. Process overviews

Figures 6.2 and 6.3 below summarise the principal foreground processes which occur at the representative farm, and during the factory stage of the supply chain. The process overview for the factory stage focuses on the production of potato products only i.e. potato crisps and what was referred to earlier as the baked ancillary product. The baked product has been included in Figure 6.2 because it shares common processes with the production of potato crisps and an understanding of these common processes will be referred to in what follows. The process overview for the distribution stage will not be further elaborated here given its simple nature, consisting solely of a dry goods warehouse operation where: 1) finished goods are temporarily stored prior to onward transit to retail customers, and 2) raw materials are stored before delivery to the factory.

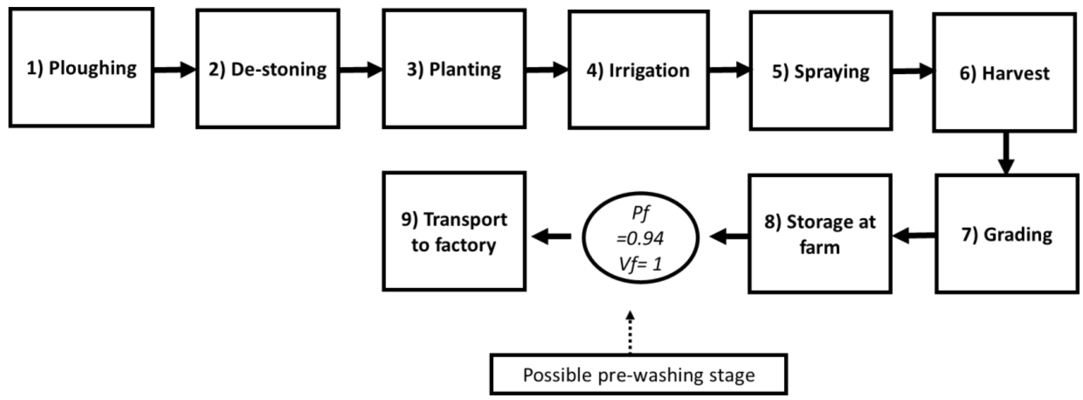

Figure 6.2. Farm stage process overview. Product (*Pf*) and value (*Vf*) fractions (see sections 6.5.4) refer to the loss of weight during storage.

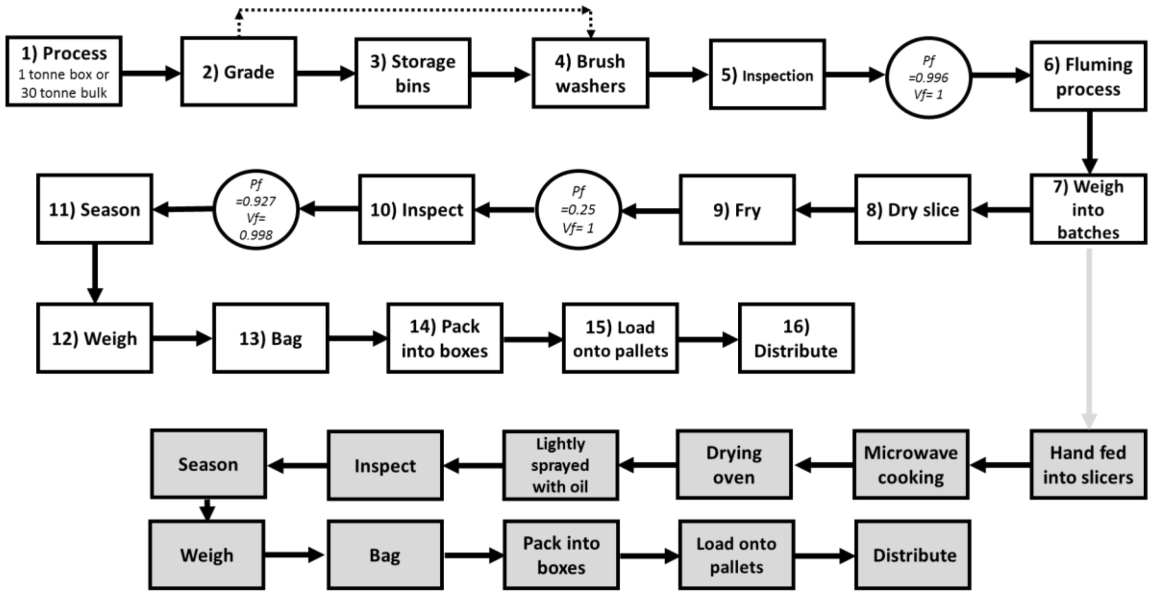

Figure 6.3. Factory stage process overview. Processes in white refer to the production of potato crisps which are the focus here. Processes in grey refer to the ancillary baked product. See section 6.5.4 for derivation of product (*Pf*) and value (*Vf*) fractions applicable to potatoes.

6.3.1 Farm stage process overview

The representative farm grows Lady Claire potatoes in four separate fields, which are planted in mid-April and harvested in late September or early October, and which together cover an area of 61 hectares. Total production is approximately 3,000 tonnes during one growing season, which equates to a yield of 49 tonnes per hectare which is in line with the average UK main crop potato yield of 45 tonnes per hectare (Nix, 2014). The potato crop is irrigated during June (40,057 m^3) and July (13,352 m^3) using a spray gun and boom, with an average irrigation interval of nine days. After harvesting, the potatoes are stored for up to nine months before they are sent to the factory. During storage, the potatoes lose approximately 6% of their weight which is mostly moisture (2,800 tonnes come ‘out of store’) which equates to a product fraction (*Pf*) at this stage of 0.94, but a value fraction (*Vf*) of 1.

In selecting between different potato varieties, the two primary crop characteristics that the company looks for relate to taste and texture. However, beyond this, the set of ideal crop characteristics - many of which are interrelated - include a dry matter content of at least 21%, the ability to produce a commercial yield and withstand storage, early maturity, low and reducing sugars, good frying colours straight out of storage, and resistance to bruising. In practice, however, each potato variety is a compromise between these various factors. For instance, in order to produce the best frying colours (golden/yellow), this necessitates sufficient time in the ground to mature and produce a commercial yield, whilst recognising that the UK potato crop has to be lifted by November otherwise there is a risk that frost could increase sugar levels and render the crop unacceptable for processing. Additionally, whilst adequate dry matter is needed in order to produce the requisite crisp texture, too high dry matters can itself increase the risk of bruising. Lady Claire, specifically, exhibits little bruising, sufficient dry matter composition, low and reducing sugar levels, consistent round tubers, good fry colours and is capable of being stored for nine months. Indeed, this ability to withstand storage makes Lady Claire particularly suitable for late season crisp production during April to June. However, it is a comparatively intensive crop requiring the best land, full irrigation and high fertiliser and spray regimes.

6.3.2 Factory stage process overview

The factory stage operates for 50 process weeks per annum (6 days per process week), producing approximately 17,500 tonnes of finished goods across all product categories including the ancillary products mentioned previously. However, as mentioned earlier, potato crisps, of multiple varieties, represent over 95% of finished goods by weight. On an annual basis, approximately 58,000 tonnes of potatoes, of all crop types, are used in the production of potato crisps and the ancillary baked product.

Key elements of the factory process overview are described in Table 6.2 below.

Table 6.2. Key elements in factory process overview

Process	Description
Grading	One to two percent (by weight) of soil is removed from the potatoes.
Grading & inspection	Five to ten tonnes of potatoes per week are rejected either because they are undersized or because they are badly damaged.
Brush washers	The potatoes are washed and any remaining dirt is removed. Annually, approximately 5,400 m^3 of soil washings (i.e. water containing suspended soil) is disposed of via local agricultural land or the sewerage system.
Fluming	A closed loop, water based, conveying system which is used to transport the potatoes during production.
Weighing	After weighing, the baked product is removed and transferred to a separate pre-cooking, cooking and finishing production line (see Figure 6.3).
Dry slice	The potatoes are cut into either a flat (as in the case here), ridge or wave profile.
Frying	During frying, on average, 75% of the moisture within the potato crop is removed leaving 3% moisture and 22% dry matter as shown in Figure 6.4 below. Note: whilst these are the typical component percentages and are thus those that will be utilised in this context, where potatoes exhibit higher dry matters and thus lower moisture content, the 3% retained moisture target remains i.e. less moisture is driven off during the frying process.
Frying	During frying, sunflower oil is added. Sunflower oil represents approximately 30% of the weight of the finished crisps that emerge from the frying process.
Inspection	Following the frying process, the crisps are inspected and it is at this point that what is known as the crisp co-product (CCP) is removed. The CCP is food safe but unsuitable for inclusion in finished goods because of, for example, imperfections such as blemishing and bruising, or, because it is too oily. The high oil content of CCP (typically around 33%) ensures that it has a high calorific value and is thus attractive to animal feed manufacturers.
Frying	The fryers used in process 9 are washed out during each process week. Typically, four tankers per week (each with a 28 tonne load) remove the water and suspended waste sunflower oil. This is processed at a local site and the waste sunflower oil is recycled for use in technical products such as lubricants as it is not suitable for use in animal feed.

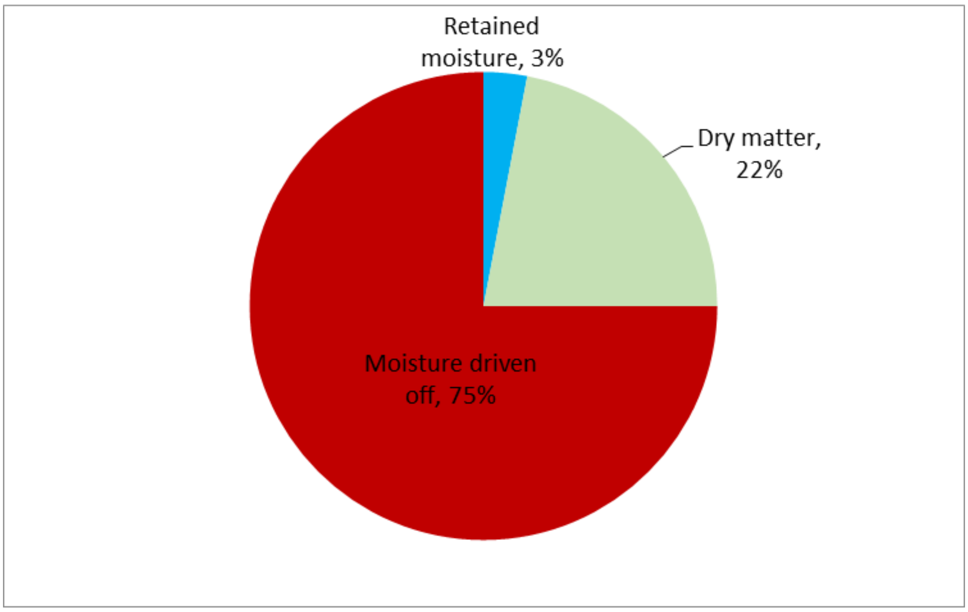

Figure 6.4. Typical potato crisp component profile.

In light of the expense that the company goes to in order to remove 75% moisture from the potato crop, it should be noted here that varieties of potato which may be resistant to reduced irrigation (i.e. which would not require as much moisture being driven off during frying) tend not to store as well, exhibit variable sugar levels and, if the dry matter content is too high, bruise far more easily. At the present time, the company suggest that there is not a crop variety which would represent both an acceptable trade-off of the desirable crop characteristics described earlier, and a substantially lower moisture content.

6.4. Annual total water withdrawal associated with potato products

Whilst the focus here is water consumption, as defined previously, Figure 6.5 below sets out the total blue water *withdrawal* figures along the supply chain focusing on the primary ingredient only. Note: figures for the farm stage are based on extrapolating the water use figures for the 2,800 tonnes of Lady Claire potatoes grown at the representative farm (measured 'out of store'), to the total annual tonnage of potatoes sourced by the company of all types (i.e. 53,410 m^3/2,800 tonnes multiplied by 58,000 tonnes = 1,106,350 m^3). Based on this assumption (which as mentioned previously is likely to be an over-estimate given the particularly intensive nature of the Lady Claire crop), in total, across the three stages of the supply chain, the annual production of potato products, of all varieties and flavours, accounts for approximately 1.17 million m^3 of water.

During the factory stage, the primary uses of the blue water withdrawals include the brush washers, which account for 55 m^3 per day, and hygiene activities over the weekend (high pressure washing, low pressure rinsing including rinsing the fryers out, hand cleaning with buckets of water, and water for rinsing down chemical residue) which account for 300 m^3. Based on 6 days per process week and 50 process weeks per year, the brush washers utilise 16,500 m^3 or approximately 28% of total factory stage withdrawals, whilst hygiene activities account for 15,000 m^3 or approximately 25%.

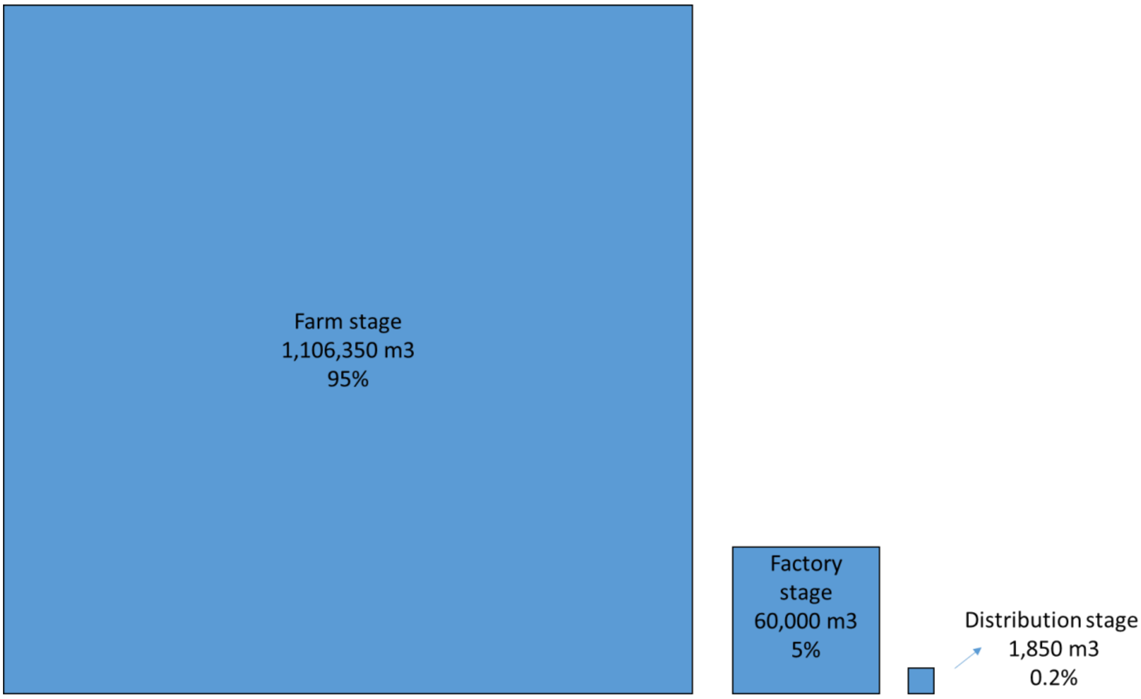

Figure 6.5. Water withdrawal volumes along the supply chain (primary ingredient only). Note: the water withdrawal volumes at the farm stage are based on extrapolating the water usage figures for the representative farm to the total annual quantity of potatoes used in the factory stage.

6.5. Supply chain water footprint directly associated with inputs

Table 6.3 below details, per ingredient/input, the quantities that go into producing a 150g bag of Salted potato crisps. In keeping with the approach adopted by Ercin *et al.* (2011), Table 6.3 also specifies the raw material underlying each ingredient/input and the origin of this raw material. The following sub-sections set out how the water footprint has been estimated for each of the three main ingredients – potato, refined sunflower oil and salt – and the raw materials which underpin the other packaging inputs.

Table 6.3. Ingredients and other inputs used to produce a 150g bag of Salted crisps

Item (section number)	Amount in grams	Raw material	Origin of raw material
Potato (6.5.1- 6.5.4)	103.5 (69%)	Potato	UK, France
Sunflower oil (6.5.5-6.5.6)	45 (30%)	Sunflower seeds	Russia, Turkey
Salt (6.5.7)	1.5 (1%)	Salt	Australia, Southern Caribbean
Plastic packet (6.5.8)	7.5	Oil	World market
Cardboard box (6.5.8)[a]	25.83	Wood	World market
Pallet (6.5.8)[b]	28.94	Wood	World market
Pallet stretch wrap (6.5.8) [c]	0.13	Oil	World market
Tape (6.5.8)[d]	0.13	Oil	World market
Pallet labels – paper (6.5.8)[f]	0.002	Wood	World market

Notes: [a] 1/12 of the 310g of cardboard used per box of 12x150g crisps. [b] 1/864 of a 25kg pallet. [c] 1/864 of 110g of pallet stretch wrap. [d] Estimate based on annual tape use of 15 tonnes. [f] Estimate based on Ercin et al. (2011).

6.5.1. Water footprint of the potato crop at Farm 1 – blue and green water consumption

The water footprint of potatoes was estimated using the methods described in Chapter Three which centre on the use of the FAO CROPWAT model which can be used to estimate blue and green water evapotranspiration (FAO, 2015b). The specific crop parameters used in the model are set out in tables A to C in Appendix 18. These reflect the standard potato profile that is built into CROPWAT, which is itself based on data from Allen *et al.* (1998). However, the potato profile was adapted, using additional data from Allen *et al.* (1998), in order to reflect the specific growth stages of a potato crop which stays in the ground for approximately 165 days, as in the of case Lady Claire here.[42]

Climate data from for use in the model, from the nearest meteorological station (in this case Gorleston, East Anglia – see Appendix 19), was sourced from the FAO CLIMWAT database which provides temperature, humidity, wind and sun data, in a format that CROPWAT can utilise (FAO, 2015a). Rainfall data, in a monthly format, was sourced directly from Farm 1. This covered the period 2006-2015 as shown in Appendix 20. The rainfall data was converted for use in the CROPWAT model using the process set out in FAO (2008) and shown in Appendix 21 together with the step-by-step results. In line with the approach adopted in Hoekstra *et al.* (2011), the USDA Soil Conservation Service method for calculating effective rainfall was adopted.

[42] The default potato profile within CROPWAT is based on a potato crop which stays in the ground for 130 days (FAO, 2015b).

Based on the above crop, climate and rainfall data, together with a yield of 49 tonnes per hectare, the Crop Water Requirement (CWR) option in CROPWAT provides the simplest means of estimating crop evapotranspiration based on a 10-day time step over the growing season. However, this does not include the water that is incorporated into the crop itself. As mentioned previously, whilst the moisture within the crop once it reaches the factory stage is typically circa 78% by weight (which equates to 0.78m^3 per tonne)[43], during nine months of storage prior to this at the farm stage, it loses approximately 6% by weight, most of which is moisture. Given this, we have added 0.84m^3 per tonne to the reference evapotranspiration estimated by CROPWAT (using the ratio of blue/green evapotranspiration estimated by the model to assign the incorporated water to the blue or green water footprint as recommended in Hoekstra *et al.* 2011). However, whether included at 78% or 84%, the water incorporated into the potatoes, in common with other crops, represents less than 1% of the total potato water footprint, in this case 0.73% (Hoekstra *et al.*, 2011). Table 6.4 and Figure 6.6 below set out the resulting green and blue water footprint associated with the potato crop at Farm 1. This is estimated for a wet, dry, normal and average year.

Table 6.4. Water footprint of potato crop production at Farm 1 (m^3/tonne) – Crop Water Requirement option

	Green	Blue	Total
Average year	63.4	51.7	115.1
Dry year	56.9	58.2	115.1
Wet year	70.0	45.2	115.1
Normal	61.5	53.6	115.1

[43] 1 litre = 1 kg or equivalently 1 m^3 = 1 tonne.

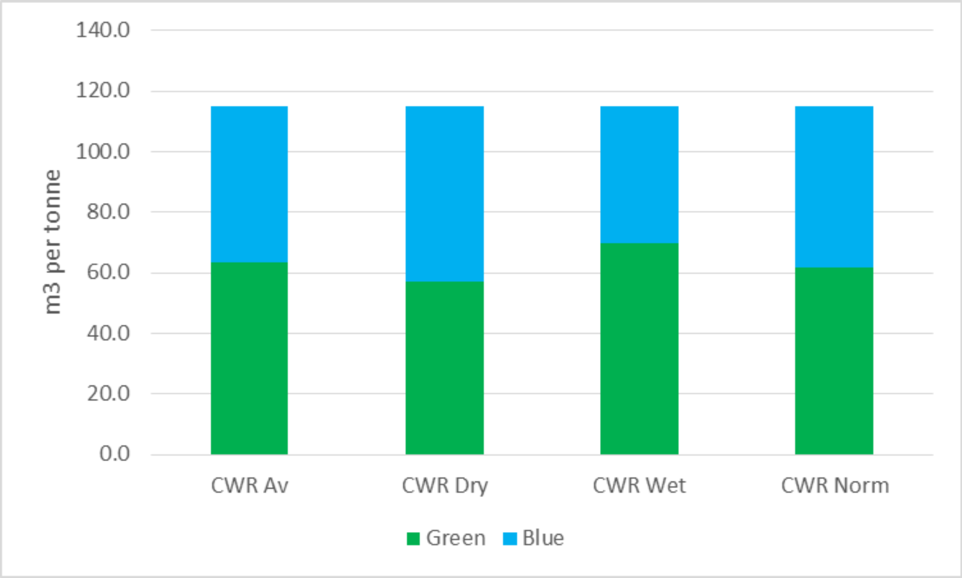

Figure 6.6. Blue and green water use in potato production at Farm 1 – Crop Water Requirement (CWR) option.

As mentioned in Chapter Three, CROPWAT also has a more accurate Irrigation Schedule (IS) option which can take account of *daily* soil moisture balance if soil data is available. Whilst detailed soil data was not obtainable for Farm 1, the medium (loam) prepopulated parameters in CROPWAT were assumed here (see Appendix 22) given that they closely resemble the anecdotal description of the soil provided (medium bodied sandy loam). Moreover, the selection of medium bodied soil is also in accordance with the approach by Hoekstra *et al.* (2011) where there is any uncertainty regarding the precise nature of the soil characteristics.

The precise method used to calculate evapotranspiration using the IS option is in line with that adopted by Mekonnen and Hoekstra (2011) and Mekonnen and Hoekstra (2010a). This involved running two scenarios in the CROPWAT model:

1. In the first scenario, irrigation was assumed to be zero (i.e. rain fed agriculture) but crop parameters were those associated with irrigated crops.
2. In the second scenario, the assumption was made that irrigation occurs and is sufficient to meet any irrigation requirement. [44]

[44] As advocated by Hoekstra et al. (2011) the irrigation parameters selected in the IS option were irrigate at critical depletion (timing) and refill soil to capacity (application) (with a field efficiency of 70%).

The green water used by the crop is assumed equal to the evapotranspiration over the growing cycle in the first scenario, whereas the blue water use is equal to the crop water use in the second scenario minus the green water use estimated in the first scenario.

Table 6.5 and Figure 6.7 below set out the green and blue water footprint associated with the potato crop at Farm 1, this time utilising the IS option. As described above for the CWR option, this includes both the reference evapotranspiration and the water that is incorporated into the crop (0.84m^3 per tonne). The results from the CWR and IS options are very similar in terms of total water consumption, however, the ratio of green and blue water is different with noticeably less blue water consumption using the more accurate IS option.[45]

Table 6.5. Water footprint of potato crop production at Farm 1 (m^3/tonne) – Irrigation Schedule option

	Green	Blue	Total
Average year	90.9	23.9	114.8
Dry year	85.7	29.1	114.8
Wet year	96.6	18.2	114.8
Normal	89.4	25.4	114.8

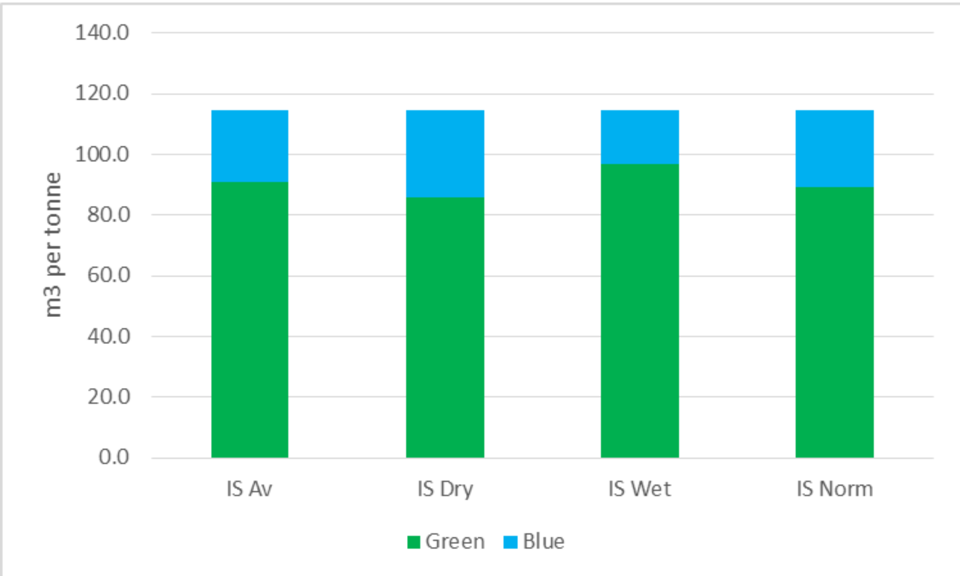

Figure 6.7. Blue and green water use in potato production at Farm 1 – Irrigation Schedule (IS) option.

[45] This is in accordance with the findings recorded of Hoekstra et al. (2011) who used the CROPWAT model to calculate the blue and green evapotranspiration associated with sugar beet cultivation.

The *normal* water footprint that was estimated for Farm 1 using the IS option will be utilised in this context henceforth as the water footprint of potatoes grown at Farm 1.[46] However, given that data was available on the irrigation applied to the Farm 1's fields (53,410 m^3), and that this is below the blue water consumption predicted by the IS option in CROPWAT (3,000 tonnes multiplied by 25.4 m^3 per tonne or 76,200 m^3), the blue water footprint at Farm 1 will be derived from actual irrigation (53,410 m^3 divided by 3,000 tonnes or 18 m^3 per tonne). This discrepancy suggests that the potato crop at Farm 1 may not be quite receiving optimal irrigation which is what the settings utilised in the CROPWAT model assume (see footnote 9), or, that the irrigation data collected perhaps refer to a wet year.

Appendix 23 presents an example output from CROPWAT (CWR option), for potato production at farm 1 during a *normal* year.[47] This has been included, in favour of an example from the IS option, because it is based on a ten-day time step and is thus comparatively brief when compared to the daily step used in the IS option.

6.5.2. Water footprint of the potato crop at Farm 1 – grey water

Franke *et al.* (2013) supplement the guidance offered in the *Water Footprint Assessment Manual* regarding the estimation of grey water using existing literature rather than sophisticated modelling approaches. This guidance can be used to estimate leaching-runoff fractions, maximum acceptable concentrations, and natural background concentrations which go beyond the more simplistic approach to the calculation of grey water adopted in the majority of the early water footprint literature (i.e. a focus on nitrogen pollution only and the utilisation of a number of simplifying assumptions which, rather than being spatially specific, are based on global averages) (e.g. Mekonnen and Hoekstra, 2011; Aldaya and Hoekstra, 2010). However, the format of the data available at Farm 1 was often not compatible with the information requirements in Franke *et al.* (2013). For example, as mentioned previously, whilst an anecdotal description of the soil type was available (medium bodied sandy loam), more specific characteristics would have been needed in order to classify the soil and thus contribute towards a bespoke leaching-runoff fraction. In addition, whilst application rates for ammonium nitrate fertiliser were available, information on how this broke down

[46] Note: any evaporative losses during irrigation, which may in some case inflate blue water consumption, have not been included here due to lack of available data.
[47] Note: refers to evapotranspiration only and does not include the water incorporated into the crop.

between NH_4 and NO_3 was not. As a result, an estimation of the maximum allowable and natural background concentrations, which differ between ammonium and nitrate, was not possible. In light of this, and so as to facilitate a fair comparison between the grey water footprint at Farm 1 and the grey water footprints of those elements of the supply chain for which secondary data, sourced from Mekonnen and Hoekstra (2010a), will be relied upon (see below), Table 6.6 sets out the grey water footprint based on the simplifying assumptions that have been used in the early water footprint literature. The grey water footprint arrived at, which is for Nitrogen fertiliser only, is based on an assumed leaching rate of 10%, natural nitrogen concentrations of zero, and a maximum acceptable concentration in the receiving water body of 10 mg/l which is in line with the drinking water standards recommended by the USA's Environmental protection Agency (see Mekonnon and Hoekstra, 2010).

Table 6.6. Grey water footprint of potato crop at farm 1 – simplified assumptions

Average N fertiliser application rate (kg/ha) [a]	Area (ha)	Total N fertiliser applied (tonne/yr)	Nitrogen leached to water bodies (tonne/yr) [b]	Maximum acceptable concentration (mg/l)	Volume of dilution water required (m^3/yr) [c]	Production (tonne/yr)	Grey water footprint (m^3/tonne) [c]
260	61	15.86	1.586	10	158,600	3,000	52.9

Notes: [a] Nitrogen fertiliser includes artificial fertiliser (ammonium nitrate) and the *contribution* from organic manure given that not all of the latter is available to the crop. [b] The load is calculated by multiplying the application rate by the area and then by the leaching rate. [c] The grey water footprint is calculated by dividing the load by the difference between the ambient water quality standards and the maximum allowable concentration (10mg/l). Note: 1.586 tonnes is equivalent to 1,586,000,000 mg. Assimilated at 10mg per litre this is requires 158,600,000 litres of dilution water, or equivalently, 158,600 m^3 [c] The grey water footprint per product is calculated by dividing the dilution volume by annual production.

The grey water footprint in Table 6.6 is above the UK average of 24 m^3 per tonne as per Mekonnen and Hoekstra (2010a). However, the application rate of nitrogen fertiliser in this context is specific to the farm in question, and the Lady Claire crop in particular, whereas the grey water footprint data estimated by Mekonnen and Hoekstra (2010a) was based on country average application rates obtained from secondary sources which assume that all potato crops grown throughout the UK require the same level of fertiliser. What is more, application rates of nitrogen fertiliser to potato crops in the UK fluctuate substantially which will have a material impact on the grey water footprint arrived at (Defra, 2014).

For completeness, Table 6.7 below presents the grey water footprint assuming leaching-run-off fractions of 1% and 25% which are the minimum and maximum fractions for nitrogen nutrients in Franke *et al.* (2013). On the assumption that natural concentrations of nitrogen are indeed zero and that the maximum allowable concentration is 10 mg/l, then the grey water footprint of potatoes at Farm 1 will lie between 5 and 132 m^3 per tonne. However, for the reason of compatibility given earlier, in this context, the grey water footprint will be referred to as that presented in Table 6.6 above which is based on what Franke *et al.* (2013) refers to as the average leaching-runoff fraction (10%).

Table 6.7. Grey water footprint of potato crop at farm 1 using minimum and maximum leaching/runoff fractions

	Average N fertiliser application rate (kg/ha)	Area (ha)	Total N fertiliser applied (tonne/yr)	Nitrogen leached to water bodies (tonne/yr)	Maximum acceptable concentration (mg/l)	Volume of dilution water required (m^3/yr)	Production (tonne/yr)	Grey water footprint (m^3/tonne)
Minimum leaching-runoff fraction (0.01)	260	61	15.86	0.1586	10	15,860	3,000	5.3
Maximum leaching-runoff fraction (0.25)	260	61	15.86	3.965	10	396,500	3,000	132.2

6.5.3. Water footprint of the potato crop at Farm 2 – blue, green and grey water

The water footprint for potato production at Farm 2 in Northern France has been taken from Mekonnen and Hoekstra (2010a). The volumetric figures correspond to the region of Nord-Pas-de-Calais within which Farm 2 is located, and represent a generic potato rather than being variety specific.

For comparison, Table 6.8 below sets out the overall water footprint of potatoes at Farm 1 (as estimated above) and Farm 2, together with the UK average potato water footprint taken from Mekonnen and Hoekstra (2010a). As mentioned, given that there is no processing of the potatoes during the farm stage, there is consequently no process water footprint to include here. It is noted that Chapagain and Orr (2010), in their study on the wheat supply chain, included the water consumed when the wheat storage house was cleaned out. However, water used for cleaning, unless it evaporates, would not represent water consumption and thus contribute towards the water footprint. Similarly, whilst the farm process overview shown in Figure 3 notes a possible pre-washing stage before the potatoes are delivered to the factory, the need for this is apparently rare and is unlikely to constitute water consumption in any case.

Table 6.8. Comparison of potato water footprints by location (m^3/tonne)

	Green	Blue	Grey	Total
Farm 1 East Anglia [a]	89.4	18	53	160.4
Farm 2 - Nord-Pas-de-Calais [b]	80	8	47	135
UK average*	66	13	24	103

Notes: [a] Normal water footprint estimated using IS option and actual irrigation data. [b] Data sourced from Mekonnen and Hoekstra (2010a).

The disparity between the grey water footprint as estimated for Farm 1, and the UK average grey water footprint, has been touched on previously. However, it is also noticeable in Table 6.8 that there is a clear difference in the estimated green and blue water footprints between Farms 1 and 2. This difference arises for two principal reasons:

1. The estimates for the potato crop at Farm 1 are based on the crop development stages in Appendix 18. These have been tailored to reflect a crop, such as Lady Claire, which stays in the ground for an extended period (around 165 days). This compares to a generic potato crop which the default profile in CROPWAT suggests reaches maturity after 130 days.
2. A comparison between the climate data used in the CROPWAT modelling here for Farm 1 (taken from the Gorleston meteorological station), and climate data taken from the two meteorological stations listed in CLIMWAT which are located in the Nord-Pas-de-Calais region, (Boulogne and Lille – see Appendix 24), is shown in Appendix 25. This indicates that Gorleston has a higher average maximum temperature (18.2 °C compared to 13 °C for Boulogne and 13.8 °C for Lille), higher average humidity (84% compared to 83% and 82%), more average sun hours (4.5 compared to 3), and crucially, higher average reference evapotranspiration (2.29 mm per day compared to 1.63 or 1.72 mm per day).

6.5.4. Product and value fractions – potatoes

The product fraction applicable to the processing of potatoes within the factory stage has been estimated based on a number of key assumptions which have been made in conjunction with the company. As will become apparent, many of these assumptions have been necessary because the data available to this study concerned the outputs from, and not inputs to, the production processes. As a result, it has been necessary in some cases to work backwards from the output values supplied in order to derive quantities of various inputs at earlier stages in the production process.

Figure 6.8 below shows the various stages of the factory process which have a bearing on the product fraction.

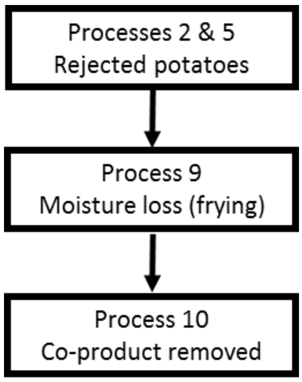

Figure 6.8. Stages of factory production process which influence the product fraction.

Given that the water footprint figures for potato cultivation estimated earlier are for the crop itself and not any soil that will remain after harvest, the weight loss associated with soil removal is not included in Figure 6.8 above and the product fraction calculations that follow. Based on an annual requirement for potatoes in 2015 of approximately 58,000 tonnes, if we assume the lower end of the range vis-à-vis the weight of soil removed during process 2 i.e. (1% by weight see Table 6.2), the 'opening balance' for the calculation of the product fraction is 57,420 tonnes. However, from this, the quantity of potatoes associated with the baked product, which leaves the crisp production process after stage 7, needs to be deducted. Table 6.9 below, working backwards from the annual quantity of baked product finished goods, details the calculations necessary to achieve this.

Note: the allocation of the weight of potatoes that enter the production process after rejected potatoes have been removed (i.e. 57,170 tonnes) between potato crisps (98.9% or 56,542 tonnes) and baked product (1.1% or 628 tonnes), is a key metric which will referred to again in what follows.

Table 6.9. Calculation of deductions from annual potato usage associated with baked product

Description	Calculation (tonnes unless specified)	Running total (tonnes)
Annual quantity of baked product finished goods		200
Assumed quantity of potato in the above		157[a]
Add back moisture removed during cooking (75%)[b]	157 /0.25	628
Add back share of potatoes rejected during processes 2 and 5 (based on 5 tonnes per week or 250 tonnes over 50 process weeks)	(1) 57,420 – 250 = 57,170 (2) 57,170 – 628 = 56,542. (3) 628 = 1.1% of 57,170; 56,542 = 98.9%. (4) 250 x 1.1% = 2.75 (5) 628 + 2.75 = 630.75	630.75
Quantity of potatoes to be deducted from 'opening balance' annual potato usage		**630.75**

Notes: [a] It was assumed that potato comprises approximately 79% of the baked product. [b] Like potato crisps, during cooking the baked product loses 75% moisture.

Drawing on Table 6.9, the product fractions associated with the processes shown in Figure 6.8, are derived in Table 6.10 below.

Table 6.10. Calculation of potato product fraction

Description	Opening balance (tonnes)	Calculation (tonnes unless specified)	Overall product fraction	Closing balance (tonnes)	Running potato product fraction [a]
Deduct potatoes used in baked product (see Table 6.9)	57,420	57,420 – 630.75	N/A	56,789.25	N/A
Deduct potatoes rejected during processes 2 and 5	56,789.25	(1) 250 x 98.9% = 247.25 (2) 56,789.25 – **247.25**	0.4%	56,542	0.996
Deduct 75% moisture during cooking	56,542	(1) 56,542 x 0.75 = 42,406.5 (2) 56,542 – **42,406.5**	74.7%	14,135.5	0.25
Deduct potato in CCP	14,135.5	(1) 1,544[b] x 0.67[c] = 1,034 (2) 14,135.5– **1,034.**	1.8%	13,101.5	0.927
Potato in finished goods	13,101.5	N/A	23.07%	13,105.5	1
Total			100%		

Notes: [a] Running potato product fraction calculated as closing balance at that stage divided by the opening balance at that stage e.g. for the CCP (13,101.5/14,135.5) = 0.927. [b] Annual quantity of CCP produced during 2015 was 1,600 tonnes. This has been pro-rated across the range of products produced by the company. [c] typical CCP composition is 33% oil and 67% potato.

Note: in Table 6.10 the *overall product fraction* shows the apportionment of the total tonnage of potatoes used in the factory between different uses. The *running potato* *product fraction*, on the other hand, illustrates the *movement* in the product fraction *between* stages.

For the associated value fractions, the potatoes that are rejected have zero value. In addition, the value fraction applicable to the CCP has been estimated based on an average value over a number of financial cycles given that it typically fluctuates quite considerably, albeit remaining a very small percentage of annual turnover. More specifically, it has been estimated as the ratio of the revenue from sales of CCP, stemming from potato crisp manufacture only, to total revenue from sales of potato crisps only. Given that CCP also contains oil, and potato crisps as a finished good also contain other ingredients, strictly speaking, the value fraction does not isolate and compare just the value of the potato in the CCP versus the value of the potato at the stage when the CCP is removed. However, it is not possible to isolate the value of the potato in the CCP as the CCP does not have a value absent the sunflower oil. Similarly, the potato used in crisp manufacture does not realise a value until it is in the form of finished goods.

Figure 6.9 below presents the running potato product fractions, and the value fractions, including the weight loss during storage at the farm stage mentioned in section 6.3.1.

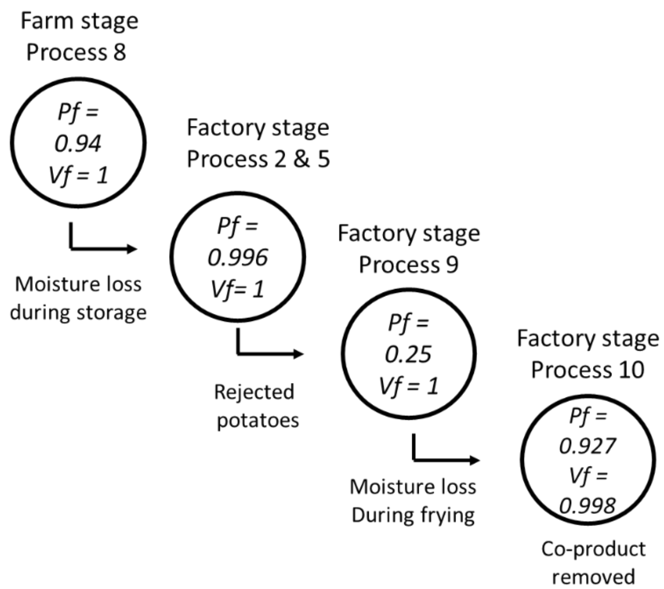

Figure 6.9. Potato product (*Pf*) and value fractions (*Vf*) along the supply chain.

In order to calculate the water footprint of the potatoes that end up in a 150g bag of Salted crisps, the water footprint figures for potatoes in Table 6.8 firstly need to be divided by the overall product fraction, and then multiplied by the value fraction. The final results of this can be seen in Table 6.11 below.

Table 6.11. Water footprint of potatoes directly used in potato crisp manufacture (m^3/tonne or litres/kg)

Location	Green	Blue	Grey	Total
Farm 1 – East Anglia	411.43	82.84	243.91	738.18
Farm 2 - Nord-Pas-de-Calais	368.17	36.82	216.30	621.28

At this point, the results in Table 6.11 are converted from litres per kg into litres per gram (Table 6.12) and then multiplied by the quantity of potato (103.5g) in a 150g bag of Salted crisps (Table 6.13). As can be seen, the final water footprint of the potatoes in a 150g bag of Salted crisps is either 76 or 64 litres depending on the origin.

Table 6.12. Water footprint of potatoes directly used in potato crisp manufacture (litres/gram)

Location	Green	Blue	Grey	Total
Farm 1 – East Anglia	0.41	0.08	0.24	0.74
Farm 2 - Nord-Pas-de-Calais	0.37	0.04	0.22	0.62

Table 6.13. Water footprint of potatoes in a 150g bag (litres)

Location	Green	Blue	Grey	Total
Farm 1 – East Anglia	42.58	8.57	25.24	76.40
Farm 2 - Nord-Pas-de-Calais	38.11	3.81	22.39	64.30

6.5.5. Water footprint of refined sunflower oil

The water footprint of refined sunflower oil, sourced from Mekonnen and Hoekstra (2010a), is presented in Table 6.14 and Figure 6.10 below. The production of refined sunflower oil involves three principal stages: first, the cultivation of sunflower seeds; second, the processing of these seeds into an unrefined or crude sunflower oil; and third, the refining of this oil into a product which is suitable for use in food production. Each of these stages are assumed to occur in the country where the sunflowers are grown (i.e. Russia or Turkey), and the process water requirements associated with each stage are included in Table 6.14 below.

Table 6.14. Water footprint of sunflower oil (refined)

	Water footprint of raw material m3/tonne [a]				Process water requirement m3/tonne [b]			
	Green	Blue	Grey	Total	Green	Blue	Grey	Total
Russia - Krasnodar Krai	7,186	27	77	7,290	0	1	0	1
Turkey - Edirne	3,221	555	436	4,212	0	1	0	1

Notes: [a] Data sourced from Mekonnen and Hoekstra (2010a). [b] Estimate sourced from data, product and value fractions in Jeffries et al. (2012).

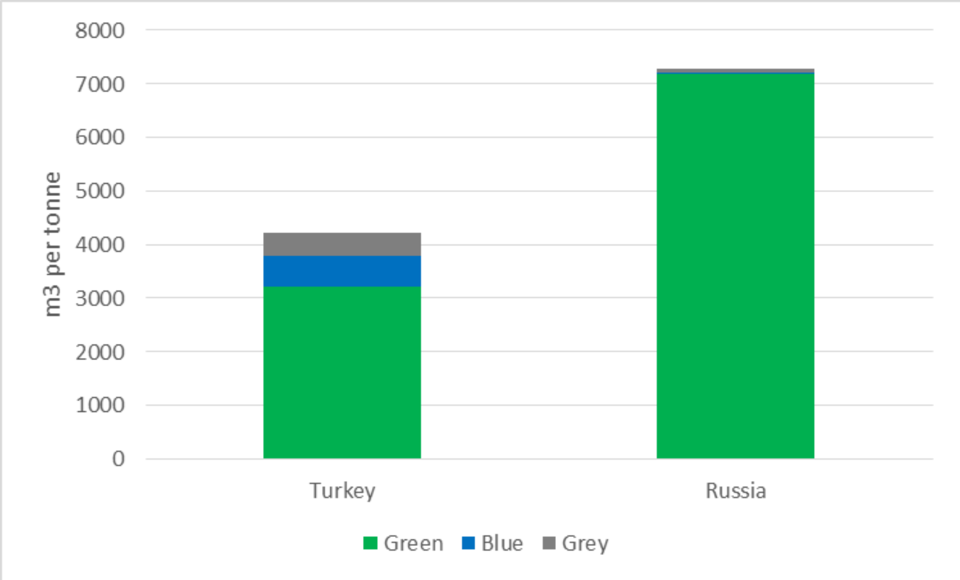

Figure 6.10. Water footprint of refined sunflower oil (as used in this study).

6.5.6. Product and value fractions – refined sunflower oil

The product fraction for sunflower oil use once it reaches the factory has been estimated as 0.823 (see Table 6.15 below). This reflects the fact that of the 4,608 tonnes of sunflower oil used per annum in the factory specifically for crisp manufacture (i.e. excluding that used in the production of the baked product), the following deductions are applicable:[48]

- 510 tonnes in the CCP[49]
- 75 tonnes as waste that is washed out of the potato crisp fryers and ultimately ends up being recovered off-site for technical uses such as lubricants

[48] Sunflower oil use in the factory has been pro-rated across the range of products produced by the company.

[49] This represents the oil content in the 1,600 tonnes of CCP which has been pro-rated across the range of products produced by the company.

- 231 tonnes which is sold on because it has deteriorated and is no longer the correct specification[50]

In terms of the value fraction of the sunflower oil that ends up in the potato crisps, this has been estimated as 0.9966 given that a) as mentioned previously the annual value of the CCP attributable to crisp manufacture represents approximately 0.2% of the value of annual potato crisp finished goods (the same rider as mentioned in section 6.5.4 also applies here), and b) the value of oil sold because it is out of specification represents approximately 0.12% of the same.

Table 6.15. Product and value fractions – sunflower oil

Description	Running total (tonnes)	Product fraction	Value fraction
Sunflower oil used in factory for potato crisp production	4,775	1	1
Deduct sunflower oil that ends up in the CCP	510	11.1%	0.22%
Deduct waste oil	75	1.6%	0
Deduct oil removed because it is no longer the correct specification	231	5.0%	0.12%
Oil used in the production of crisps	**3,959**	**82.3%**	**99.66%**

In order to calculate the water footprint of the sunflower oil that ends up in a 150g bag of Salted crisps, the water footprint figures for refined sunflower oil in Table 6.14 firstly need to be divided by the product fraction and then multiplied by the value fraction. The results of this can be seen in Table 6.16 below.

Table 6.16. Water footprint of sunflower oil directly used in potato crisp manufacture (m^3/tonne or litres/kg)

Location	Green	Blue	Grey	Total
Russia - Krasnodar Krai	8,701.73	33.91	93.24	8,828.93
Turkey - Edirne	3,900.42	673.28	527.97	5,101.67

At this point, the results in Table 6.16 are converted from litres per kg into litres per gram (Table 6.17) and then multiplied by the quantity of sunflower oil (45g) in a 150g bag of Salted crisps (Table 6.18). As can be seen, the final water footprint of the sunflower oil in a 150g bag of Salted crisps is either 228 or 394 litres depending on the origin.

[50] Waste and out of specification oil are almost exclusively associated with potato crisp production only, to the extent that no meaningful apportionment of these two items could be made.

Table 6.17. Water footprint of sunflower oil directly used in potato crisp manufacture (litres/gram)

Location	Green	Blue	Grey	Total
Russia - Krasnodar Krai	8.70	0.03	0.09	8.83
Turkey - Edirne	3.90	0.67	0.53	5.10

Table 6.18. Water footprint of sunflower oil used in a 150g bag (litres)

Location	Green	Blue	Grey	Total
Russia - Krasnodar Krai	391.58	1.53	4.20	397.30
Turkey - Edirne	175.52	30.30	23.76	229.58

6.5.7. Water footprint of salt

The water associated with the 1.5 grams of salt that is present in the 150g bag of Salted crisps has been excluded from the analysis here. This is in line with the approach adopted by Aldaya and Hoekstra (2010) who likewise excluded the water related to salt in their analysis of the pasta water footprint because it was not deemed significant from a water perspective. Indeed, following a review of the unit process datasets that are contained in Ecoinvent (2013) regarding sodium chloride production (powder) using solution mining (which would appear to be the most water intensive salt production method), in which the water consumed was estimated as the water taken from the environment minus the water returned to the environment, it appears that only approximately 2 litres of water is consumed per kg of salt. As a result, the 1.5 grams of salt in the bag of crisps would only account for approximately 2.5 millilitres of water consumed and therefore is rightly excluded here as it lacks significance in this context.

6.5.8. Water footprint of other inputs

Table 6.19 below presents the water footprints and process water requirements associated with the raw materials which underpin the potato crisp packaging inputs. As indicated, the data in Table 6.19 closely follows the source data cited by Ercin *et al.* (2011) although it has been augmented with data from the 3rd edition of *The Water Encyclopaedia* (Fierro and Nyer, 2011) where possible. The water footprint for each item is based on what is an assumed location in order to facilitate the analysis i.e. it was not possible to gain full visibility over where the raw materials originated from. Strictly speaking, the figures in Table 6.19 do not refer to water consumption, but rather the *water required per unit*, and as such could be considered an overestimate in this context. However, accounting for the water burden associated with the raw material underpinning the item, and not the item itself, introduces an element of conservatism here to counterbalance this. For example, the production of one tonne of paper products will likely require more than one tonne of wood as an input. Overall, the figures represent the best estimate given available data.

Table 6.19. Water footprint of packaging input raw materials

Item	Raw material	Selected location	Water footprint m^3/tonne of raw material			Process water requirement m^3/tonne		
			Green	Blue	Grey	Green	Blue	Grey
Plastic packet	Oil	Sweden (raw) German (process)	0	10[b]	0	0	0	225[b]
Cardboard box	Wood	Finland	369.4[a]	0	0	0	0	125[c]
Pallet	Wood	Sweden (process)	369.4[a]	0	0	0	0	75[b]
Pallet stretch wrap	Oil	Sweden (raw) German (process)	0	10[b]	0	0	0	225[b]
Tape	Oil	Sweden (raw) German (process)	0	10[b]	0	0	0	225[b]
Pallet labels	Wood	Sweden (process)	369.4[a]	0	0	0	0	500[c]

Notes: [a] Data sourced from Gerbens-Leenes et al. (2009) *cited in* Ercin et al. (2011). [b] Data sourced from Van der Leeden et al. (1990) *cited in* Ercin et al. (2011) [c] Data sourced from Fierro and Nyer (2011).

Based on the water footprint data in Table 6.19, together with the quantities of packaging inputs that are applicable to a 150g bag of crisps noted in Table 6.3, Table 6.20 below derives a water footprint of *other inputs* of 27.46 litres, over 90% of which is associated with the cardboard box and wooden pallet.

Table 6.20. Water footprint of packaging inputs used for a150g bag of crisps

Item	Total water footprint in litres (raw material and process water)				%
	Green	Blue	Grey	Total	
Plastic packet	0.00	0.08	1.69	1.76	6
Cardboard box	9.54	0.00	3.23	12.77	47
Pallet	10.69	0.00	2.17	12.86	47
Pallet Stretch wrap	0.00	0.00	0.03	0.03	<1
Tape	0.00	0.00	0.03	0.03	<1
Pallet labels	0.00	0.00	0.00	0.00	<1
Total	20.23	0.08	7.15	27.46	100

6.6. Operational water footprint directly associated with inputs

As referred to in Chapters Two and Three, the operational water footprint refers to water that is consumed (blue) or degraded (grey) during the production of potato crisps in the factory stage. Water consumption is defined as (Hoekstra *et al.* 2011):

- Water that evaporates.
- Water that is incorporated into the product.
- Water that is returned to a different catchment area.
- Water that is not returned in the same period.

In terms of the first two categories, as mentioned, the potato crisp production process is focused on removing moisture from the potatoes (75% by weight which is equivalent to 0.75 m^3 per tonne) which is lost as evaporation during cooking.[51] However, this cannot be included as water consumption here because the cooking process is simply removing water that was previously consumed during potato cultivation i.e. to include it here as well would be double-counting. More broadly, water does not evaporate at any other point during the process overview depicted in Figure 6.3.

There are two principal instances of the spatial displacement of water:

- Annually, 5,400 m^3 of soil washings (i.e. the water used to wash the potatoes in the brush washers which contains soil residue) are created. A portion of this is recycled to local agricultural land away from the factory.
- The water that is used to wash out the fryers, and which contains suspended waste sunflower oil, is processed at a plant at a separate location.

However, in both instances, the spatial displacement does not stretch beyond the river basin where the company is located, and as a result, neither activity constitutes water consumption.

Overall, the operational water footprint is zero. Similarly, all of the wastewater produced during the potato crisp production process, with the exception of the two instances of spatial displacement noted above, is returned via a public sewage system to a waste water treatment plant. In conjunction with the fact that soil washings don not constitute a pollutant and that all suspended waste sunflower is removed and processed, the grey water footprint here is also assumed to be zero.

6.7. Supply chain overhead water footprint

In line with Ercin *et al.* (2011) and Jeffries *et al.* (2012), only the water footprints associated with certain key items used in the factory and distribution stages, but not

[51] Assuming 56,542 tonnes of potatoes are used in potato crisp production per year (after rejected potatoes), evaporation amounts to 42,406 m^3.

directly linked to production, are considered here as shown in Table 6.21 below. For concrete, steel and vehicles, total amounts are presented in conjunction with a lifespan in years which can be used to amortize the water burden on an annual basis.

Table 6.21. Supply chain overhead water footprint – items selected for analysis

	Total amount used	Unit	Raw material	Amount of raw material	Unit of raw material	Lifespan of raw material	Yearly amount
Concrete[a]	3,840	Tonnes	Cement	3,840	Tonnes	40	96
Steel[b]	640	Tonnes	Steel	640	Tonnes	40	16
Paper	3.25	Tonnes/year	Wood	3.25	Tonnes/year	-	3.25
Natural[c] Gas	172,800	GJ/Year	Gas	172,800	GJ/Year	-	172,800
Electricity[d]	24,120	GJ/Year	Several	24,120	GJ/Year	-	24,120
Vehicles[e]	45	Numbers	Steel	10	Tonnes/vehicle	10	45
Fuel[f]	155,000	Litres/yr	Diesel	155,000	Litres/yr	-	155,000

Notes: [a] Estimate of concrete usage is based on the ratio between concrete and steel used in Ercin et al. (2011). [b] Steel usage has been estimated as 4kg per square foot (total square footage of factory and distribution facilities is 160,000 sq. ft). [c] GJ per year equivalent to 48 m/kwh (natural gas is only used in the factory stage). [d] GJ per year equivalent to 6.7 m/kwh (represents electricity use in the factory and distribution stages). [e] Conservative estimate of average steel per vehicle across nine tractor units, 32 trailers and four fork lift trucks. [f] Diesel fuel accounted for is the annual amount associated with haulage journeys between factory and distribution centre.

Table 6.22 below presents the water footprints of the raw materials which underpin the overhead items, together with the process water requirements. Note: the figures presented closely follow those used by Ercin *et al.* (2011) although they have been

Table 6.22. Supply chain overhead water footprint – raw material estimates

	Raw material	Selected location	Water footprint m^3/ tonne of raw material			Process water requirement m^3/tonne (unless indicated)		
			Green	Blue	Grey	Green	Blue	Grey
Concrete	Cement	Global average	0	0	0	0	0	1.46[a]
Steel	Steel	Sweden (process) USA (raw material	0	4.2[c]	0	0	0	61[c]
Paper	Wood	Sweden (process)	369.4[b]	0	0	0	0	500[d]
Natural Gas (per GJ)	Gas	World average	0	0	0	0	0	0.11[c]
Electricity (per GJ)	Several	World average	0	0	0	0	0	0.47[c]
Vehicles	Steel	Sweden (process) USA (raw material	0	4.2[c]	0	0	0	61[c]
Fuel (per m^3)	Diesel	USA	0	0	0	0	0	8.5[e]

Notes: [a] Average of process water requirements in Belgium, Cyrpus (dry process), Finland, USA (wet process) (source: Fierro and Nyer, 2011). [b] Data sourced from Gerbens-Leenes et al. (2009) *cited in* Ercin et al. (2011). [c] Data sourced from Van der Leeden et al. (1990) *cited in* Ercin et al. (2011). [d] Data sourced from Fierro and Nyer (2011). [e] Mid-range estimate based on gasoline production in the USA (source: Fierro and Nyer, 2011).

augmented where possible using data from Fierro and Nyer (2011). In common with the figures for water use associated with *other inputs* noted in section 6.5.8 above, strictly speaking, the figures in Table 6.22 do not refer to water consumption, but rather the *water required per unit*. Indeed, the same riders as noted in section 6.5.8 apply here.

Based on the yearly amounts of the items selected for analysis in Table 6.21, and the raw material water footprint figures in Table 6.22, the *total* annual supply chain overhead water footprint is approximately 38,600 m^3 as set out in Table 6.23 below.

Table 6.23. Total supply chain overhead water footprint

Raw material	Total water footprint in m^3 (raw material and process water)				%
	Green	Blue	Grey	Total	
Cement	0	0	140.16	140.16	<1
Steel	0	67.2	976	1,043.20	3
Paper	1,200.55	0	1,625	2,825.55	7
Natural gas	0	0	19,008	19,008.00	49
Electricity	0	0	11,336.4	11,336.40	29
Vehicles	0	189	2,745	2,934.00	8
Diesel	0	0	1,317.5	1,317.50	3
Total	1,200.55	256.20	37,148.06	38,604.81	100

The water burden in Table 6.23 is assigned to 150g bags of Salted potato crisps based on the ratio of the annual production value of the product to the annual value of all products produced at the factory. We have estimated the ratio as 14% given that a) Salted potato crisps represent approximately 14% of the weight of all finished goods (2,400 tonnes), and b) assuming a constant sales price per tonne across product categories. Per annum, approximately 16 million bags of Salted potato crisps are produced, so this fraction (1/16m) of the supply chain overhead water footprint applicable to Salted crisps (5,405 m^3), is allocated to each bag, equating to approximately 0.34 litres (0.01 litres green water, 0.002 litres blue water, 0.33 litres grey water).

6.8. Operational overhead water footprint

Given the commentary in Section 6.6 above regarding the ultimate destination of waste water associated with the factory (i.e. it is returned via the public sewerage system to a water treatment plant), it is assumed that the water used in the toilets, and for hygiene and cleaning activities in the factory, does lead to a grey water footprint. However, we have assumed that each employee drinks on average 1 litre of water per day, and in line with the approach adopted by Jeffries *et al.* (2012), that 35% of this is evaporated through breathing and perspiration and thus represents blue water consumption. Table

6.24 below presents the volumes of blue water associated with drinking water consumption by employees based on these assumptions.

Table 6.24. Blue water consumption from drinking water

Site	Average number of employees in a 24 hour period	Assumed daily water intake (litres)	Annual water intake based on 6 process days per week (litres)	Water evaporation (35%) (litres)	Water evaporation (35%) m^3
Stage 2 Factory	210	1	65,520	22,932	22.9
Stage 3 Distribution	30	1	9,360	3,276	3.3
Total	240	1	74,880	26,208	26.2

As per section 6.7 above, the 26.2 m^3 in Table 6.24 is assigned to 150g bags of Salted potato crisps based on the ratio of the annual production value of the product to the annual value of all products produced at the factory (estimated at 14% which equates to 3.7 m^3). It is recognised that this is a small volume of water, particularly when amortized over the 16 million Salted 150g bags sold (less than one millilitre per bag). Indeed, the operational overhead water footprint could easily be excluded on the basis of materiality. However, it is retained here in line with other water footprint studies which try, where possible, to estimate the water consumption within the main factory stages of product supply chains.

6.9. Total water footprint

Drawing on sections 6.5-6.8 (and in particular tables 6.13, 6.18, 6.20 and 6.23), the total water footprint of a 150g bag of Salted potato crisps can now be derived and is set out in Table 6.25 below. As can be seen, the water footprint varies between 322 litres and 502 litres depending on the origin of the potatoes and sunflower oil.

Table 6.25. Total water footprint of 150g bag of Salted potato crisps (litres)

Water footprint (litres)	Potatoes - UK (East Anglia)				Potatoes - France (Nord Pas-de-Calais)			
	Green	Blue	Grey	Total	Green	Blue	Grey	Total
Sunflower oil - Russia (Krasnodar Krai)	454	10	37	502	450	5	34	489
Sunflower oil - Turkey (Edirne)	238	39	57	334	234	34	54	322

Notes: Water footprint figures calculated by varying origin of potatoes and sunflower only. All other water footprint components are held constant. Figures correct to the nearest whole unit.

In common with other studies of a similar type, the vast majority of the water footprint of potato crisps is related to supply chain inputs which are directly associated with production (section 6.5), as shown in Table 6.26 below.

Table 6.26. Composition of total water footprint

Item (section number)	Water footprint in litres				
	Green	Blue	Grey	Total	% of total
Potato cultivation [a] (6.5.1 – 6.5.4)	42.58-38.11	8.57-3.81	25.24-22.39	76.40-64.30	13-23
Sunflower cultivation [a] (6.5.5-6.5.6)	391.58-175.52	30.30-1.53	23.76-4.20	397.30-229.58	69-81
Salt (6.5.7)					
Other inputs (6.5.8)	20.23	0.08	7.15	27.46	6-9
Operational (6.6)	0	0	0	0	0
Supply chain overhead (6.7)	0.01	0.002	0.33	0.34	0.068-0.106
Operational overhead (6.8)	0	0.0002	0	0.0002	0

Notes: [a] The range of water footprint values arises because more than one origin was studied. Columns should not be added as highest values in each range do not refer to the same country in every instance.

What is particularly noticeable in Table 6.25 and 26 is both the volume of water used in sunflower cultivation (representing between 69 and 81% of the total water footprint depending on the origin of potatoes and sunflower oil) and the wide disparity in the composition of this water usage. The latter can be seen more clearly in Figure 6.11 below which shows the *total* water footprint of potato crisps according to the four possible combinations of origin of potatoes and sunflower oil.

The presence of Turkey, quite clearly, leads to inflated blue and grey water footprints given that sunflower oil production in Edirne requires approximately 20 times more surface and/or groundwater than in Krasnodar Krai, and approximately 6 times more grey water. Nonetheless, *total* water usage in Turkey is less than that in Russia given the very large volumes of green water used in the latter (123% more green water is used in Russia compared to Turkey). Indeed, the volume of green water use in Russia is greater than the total sunflower oil water footprint in Turkey.

Lady Claire potatoes sourced in the UK represent 15% or 23% of the total water footprint depending on whether the sunflower oil is sourced from Russia or Turkey, compared to 13% or 20% when potatoes are sourced from France. This reflects the fact that, based on the assumptions made and in particular the secondary data that was relied on for the generic French potato crop, Lady Claire potatoes grown in the UK have a water footprint approximately 19% bigger than their French equivalent. As modelled, UK potatoes consume 2.25 times the blue water, and approximately 12% more green and grey water, when compared to the French crop. As mentioned previously, this reflects the growth cycle of the Lady Claire potato when compared to the generic potato type in France, differing climatic conditions, and it should be added, differences stemming from the comparison of more accurate primary data for the UK with secondary data for France.

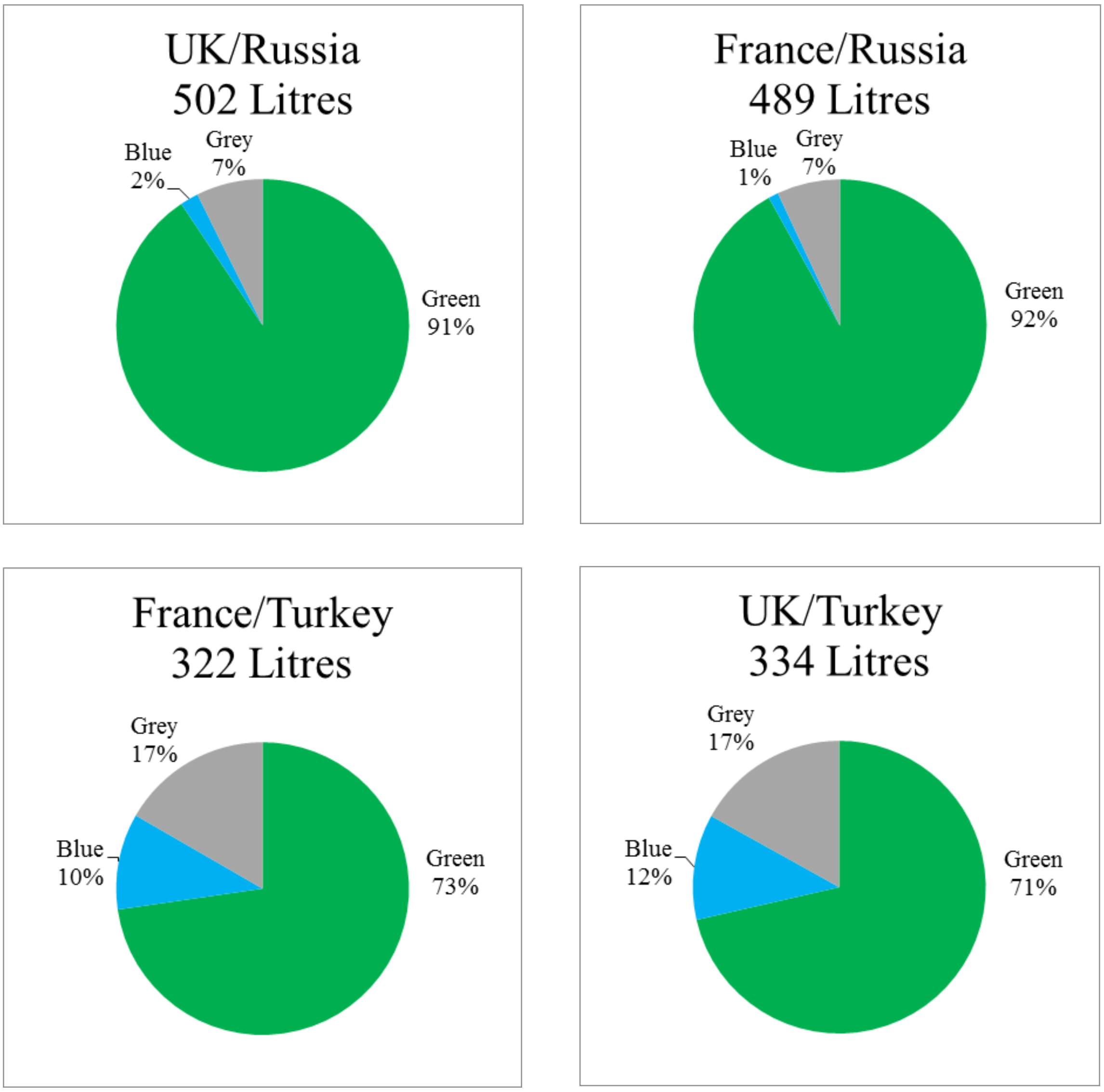

Figure 6.11. Total water footprint according to origin of potatoes and sunflower oil.

For completeness, Table 6.27 below presents the water footprint of one tonne of Salted potato crisps, this time measured in cubic metres as opposed to litres. As mentioned, this is included here in order to provide more meaningful units of analysis when it comes to the valuation of these water flows, monetary estimates of which tend only to register in higher volumetric measures (note: as with the tea case study in Chapter Five, linear aggregation has been assumed here with no economies of scale).

Table 6.27. Total water footprint of one tonne of Salted potato crisps (6,667 150g bags)

Water footprint (m^3)	Potatoes - UK (East Anglia)				Potatoes - France (Nord Pas-de-Calais)			
	Green	Blue	Grey	Total	Green	Blue	Grey	Total
Sunflower oil - Russia (Krasnodar Krai)	3,030	68	246	3,344	3,000	36	227	3,263
Sunflower oil - Turkey (Edirne)	1,589	260	377	2,225	1,559	228	357	2,145

Notes: Water footprint figures calculated by varying origin of potatoes and sunflower only. All other water footprint components are held constant. Figures correct to the nearest whole unit.

Figures 6.12 and 6.13 below provide a visual representation of the geographical distribution of the water consumed in the production of one tonne of potato crisps for both the lowest (France and Turkey) and highest (UK and Russia) combinations of potato and sunflower oil sourcing locations. Note: as referred to in Chapters Four and Five, for those aspects of the water footprint which are not geographically specific (packaging inputs and the supply chain overhead water footprint), the water burden is assumed to fall where these items are used (i.e. the factory stages).

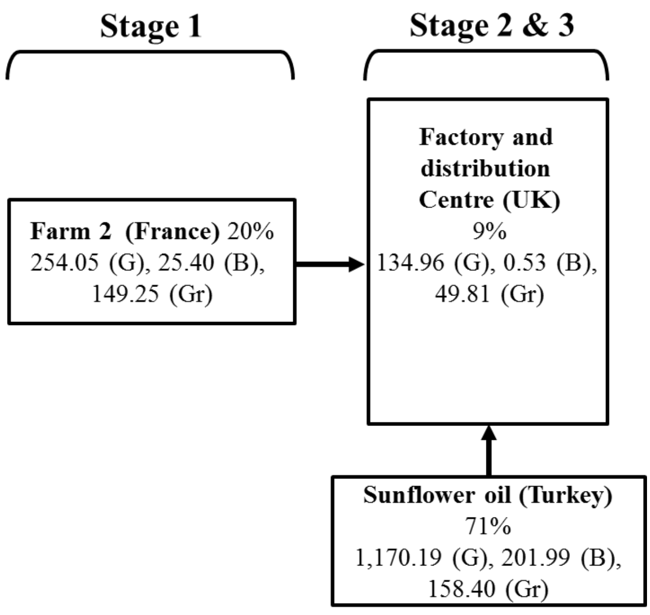

Figure 6.12. Water footprint associated with one tonne of potato crisps (low scenario).

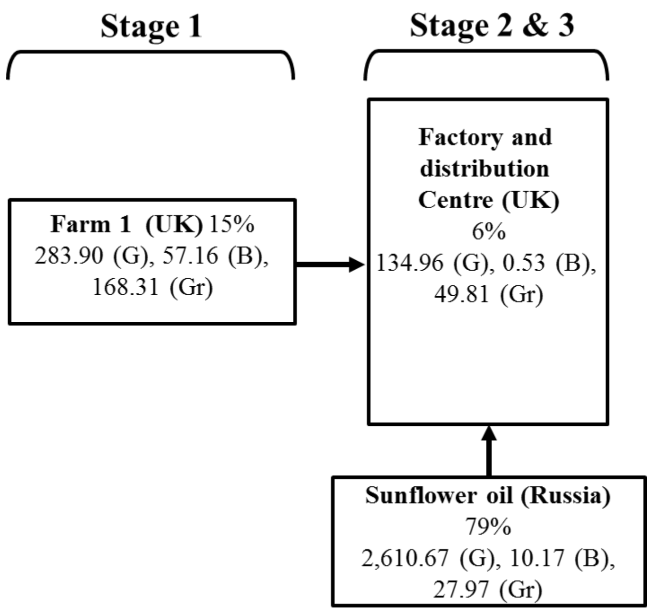

Figure 6.13. Water footprint associated with one tonne of potato crisps (high scenario).

If the volumetric water footprint alone was being used for decision making, it is quite clear that potatoes and sunflower oil sourced from France and Turkey, respectively, would consume and degrade the lowest *total* volume of water. However, water footprinting also takes in to account the vulnerability of water systems using the water stress index which measures the ratio of total annual water withdrawals in an area to total annual water availability. Table 6.28 below sets out the water stress values for each of the sourcing locations at stage 1. These can be used to assess the impact of *blue water* usage in the supply chain and thus identify 'hotspots.'[52] Following the approach set out in Jeffries *et al.* (2012, p.159), a hotspot occurs where 'the blue water footprint of products is large and where water scarcity is high,' the latter being defined as where it exceeds a value 0.6. In this context, this would suggest that Edirne, which exhibits a water stress value of 0.66 and is responsible for substantial quantities of blue water consumption in the production of sunflower oil (555 m^3/tonne), which itself represents the largest component of the overall water footprint (Table 6.26), is a hotspot location. In addition, although to a lesser extent, potato production at Farm 2 in Nord-Pas-de-Calais exhibits a water stress value of 0.65 and clearly evident blue water volumes. As a result, based on an analysis of blue water and its scarcity, the conclusion arrived at based on volume alone may be reversed as this suggests that sourcing potatoes from the

[52] The water stress index does not measure the sustainability of the green water footprint which, Jeffries et al. (2012) suggest, is a largely unexplored field.

UK (water stress value of 0.41) and sunflower oil from Russia (water stress value of 0.08 and predominantly rain-fed), may be the optimum combination. However, choices such as these may be aided by a focus on the monetary valuation of these water volumes, a subject to which Part B now turns.

Table 6.28. Baseline water stress values

Location (crop)	Baseline water stress
UK - East Anglia (potatoes)	0.41
France - Nord-Pas-de-Calais (potatoes)	0.65
Turkey - Edirne (sunflower oil)	0.66
Russia - Krasnodar Krai (sunflower oil)	0.08

Source: World Resources Institute (2013).

Part B – Unit water values along the supply chain

Having looked at the *volumes* of water that are consumed and degraded in the potato crisp supply chain in Part A, Part B now turns to the *monetary value* of these volumes of water and what this can add to water footprint assessment. As with the pasta (Chapter Four) and tea (Chapter Five) case studies previously, Part B begins (section 6.10) by looking at the value of the blue water consumed in the supply chain, followed in sections 6.11, by the value of the grey water that is degraded. Following this, section 6.12 will comment on the suitability of valuing green water according to the method set out in Chapter Three. Again, the focus here throughout will be direct use value i.e. the value of off-stream extractive water use. However, unlike the tea and pasta case studies, as we have seen there is no consumer use phase in the potato crisp supply chain, and there is no substantial consumptive and geographically specific water footprint at the factory stages. Consequently, the focus of the analysis here will be the water consumed in the agricultural stage during the cultivation of potatoes (UK and France) and sunflower oil (Russia and Turkey). Nevertheless, given that this case study has estimated the volumes of water *withdrawn* along the supply chain (section 6.4), it will also estimate water withdrawal values (section 6.14) something which was not possible using the secondary data in the tea and pasta case studies.

6.10 Blue water value

The unit values of blue water used in the production of potatoes and sunflower oil that have been selected for use in this context are set out in Table 6.29 and 6.30 below.[53] As

[53] Note: Unit values are reported throughout at two decimal places. However, any calculations that make use of the unit values have been carried out using more exact figures where these were available.

Table 6.29. Agricultural values used in the potato supply chain (potatoes)

Supply chain location at Stage 1 (Policy site)	Source	Method	Value type	At site/ at source	Long run/short run	Water volume measure	Crop value	Original value m^3 (currency)	2014 USD per m^3 *	Study location (Study site)
Farm 1 East Anglia (UK)	Knox et al. (2000)	Yield comparison	AV	Unclear	Long	Application	Potatoes	1.76 GBP	3.68	East Anglia - UK
Farm 2 Nord-Pas-de-Calais (France)	Knox et al. (2000)	Yield comparison	AV	Unclear	Long	Application	Potatoes	1.76 GBP	3.68	East Anglia - UK

Notes: * Values converted from local currency to 2014 USD using World Bank PPP exchange rates and Implicit Price Deflator (Appendix 3). See Chapter Three.

Table 6.30. Agricultural values used in the potato supply chain (sunflower oil)

Supply chain location at Stage 1 (Policy site)	Source	Method	Value type	At site/ at source	Long run/short run	Water volume measure	Crop value	Original value m^3 (currency)	2014 USD per m^3 *	Study location (Study site)
Russia (Krasnodar Krai)	Hellegers & Perry (2004)	Farm crop budget	AV	At site	Short	Application	Low (multiple – unclear)	0.11 (USD)	0.13	Crimea
Turkey (Edirne)	Latinopoulos et al. (2004)	Hedonic	MV	At site	Long	Withdrawal	Unclear	0.06 (Euros)	0.12	Chalkidiki - Greece

Notes: * Values converted from local currency to 2014 USD using World Bank PPP exchange rates and Implicit Price Deflator (Appendix 3). See Chapter Three.

shown, the value of water used in potato cultivation in the UK and France has been taken from a single source – Knox *et al.* (2000) – which is specific to *main crop* potato production in East Anglia in the UK. Whilst there is obviously a strong correlation between both the crop type, and study and policy sites, for potatoes produced at Farm 1 in the UK, owing to the absence of values for potato production in France, it has been assumed here that the values which prevail in the UK also prevail at Farm 2 in France. This is not an unreasonable assumption given the similarly advanced and proximate nature of the respective economies involved, but nevertheless, it is a limitation which means that it will not be possible to assign different unit values to potatoes produced in the two locations. As a result, the conclusions which will be drawn vis-à-vis potato sourcing locations, will be driven by volume differences alone. However, a number of sensitivities will also be incorporated in what follows in order to assess how any conclusions might change with variations in the unit value of water in France.

A word of caution is appropriate here about the monetary values from Knox *et al.* (2000) that have been utilised in this context. The authors use what is best described as a yield comparison type approach in that they look to 'estimate the combined increased yield and quality assurance benefits that irrigation would provide to the farmer, assuming that the farmer would grow the same crops with or without irrigation (Knox *et al.*, 2000, p.49). However, whilst the authors allow for *additional* crop production costs, variable costs and fixed costs (i.e. the values estimated are not a crude estimate of value similar to what was referred to in Chapter Three as *gross value* estimates), the *financial benefits* per m^3 that they arrive at, may be an overestimate depending on the precise costs allowed for which are not fully clear. Ideally, a net value of water, allowing for all costs, would have been estimated before and after irrigation in order to isolate just the value of just the irrigation water. However, as mentioned above, the values in Knox *et al.* (2000) are specific to potatoes and the policy site at Farm 1 and thus represent the best data available in this context. In addition, as with all the data in tables 6.29 and 6.30 below, the values for blue water are for application or withdrawal, and as such, represent a lower bound estimate of the value of water consumption in this context and are thus more defensible.

As regards sunflower oil, the value for Krasnodar in Russia has been taken from Hellegers and Perry (2004) who estimate the value of irrigation water used in low valued crops in the Crimean region which is directly adjacent to Krasnodar. The value of blue water in Turkey has been taken from Latinopoulos *et al.* (2004) who estimate the value of irrigation water for unspecified crops in a rural region of northern Greece which is directly proximate to Edirne in northern Turkey. It should be noted here that the value for potatoes is specific to that crop type, whereas the value assigned to sunflowers, which is itself a low valued crop, comes from either multiple low valued crops (Hellegers and Perry, 2004) or unspecified crops (Latinopoulos *et al.* 2004) which will include multiple low valued crops. As a result, direct comparisons between the *relative* value of water in potato and sunflower cultivation, whilst inevitable, should be treated with caution here. In addition, strictly speaking, the values estimated by Latinopoulos *et al.* (2004) using the hedonic method are *marginal* values, and as such, are not directly comparable with the *average* values estimated by the other authors in tables 6.29 and 6.30. However, tables 6.29 and 6.30 represent the best available data in this context (indeed the sources listed are the only ones available for the geographies in question), and as with unit values for potato production in France, the unit value of water use assigned to sunflower oil production in Turkey will also be sensitised in what follows.

Figure 6.14 (low scenario) and Figure 6.15 (high scenario) set out the unit values that have been assigned to blue water consumption at each stage along the potato crisp supply chain, together with the value of the specific volume of blue water used at each stage (based on one tonne of potato crisps). As referred to previously, there is no substantial geographically specific blue water footprint associated with factory stages 2 and 3.

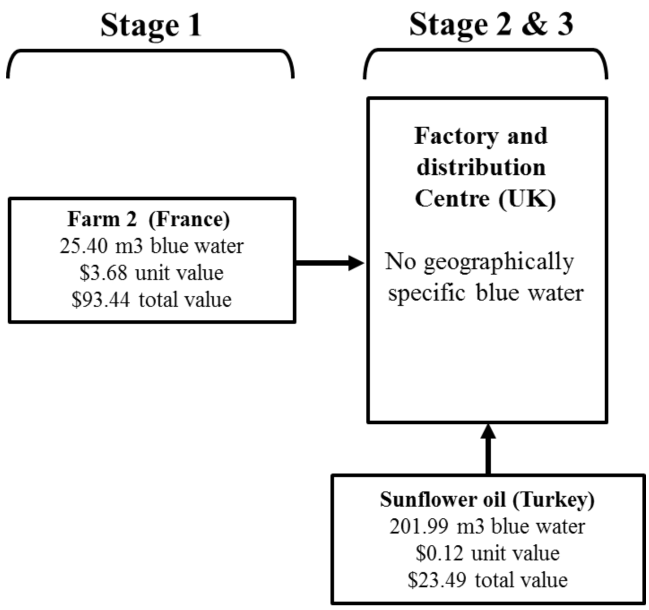

Figure 6.14. Blue water values assigned to each stage of the potato crisp supply chain (low scenario).

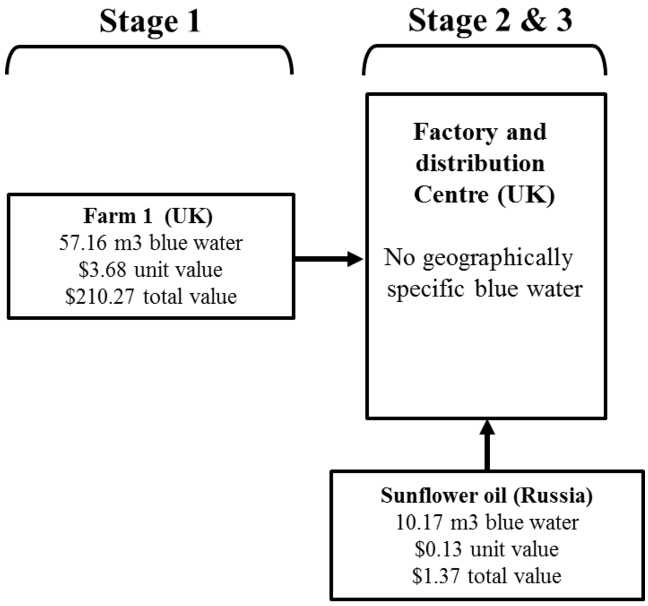

Figure 6.15. Blue water values assigned to each stage of the potato crisp supply chain (high scenario).

As shown in Table 6.31 below, in the low scenario, water consumption in potato cultivation accounts for only 11% by volume but 80% by value given the vastly different unit values noted in tables 6.29 and 6.30 above. The total direct use value of blue water consumed in the production of one tonne of potato crisps (low scenario) is estimated at $117, or, using the prevailing nominal exchange rate in mid 2017 (1 USD = 0.77 GBP), approximately £90.

Table 6.31. Value and volume of blue water used to produce one tonne of potato crisps (low scenario)

Crop (location)	Volume of blue water (m^3)	Unit value (USD 2014)	Value of blue water consumed (USD 2014)	% of total blue water volume	% of total blue water value
Potatoes (Nord-Pas-de-Calais - France)	25.40	3.68	93.44	11	80
Sunflower oil (Edirne - Turkey)	201.99	0.12	23.49	89	20
Total	227.39		116.93	100	100

In the high scenario shown in Table 6.32, water consumption in potato production accounts for a much higher share (85%) of total blue water volumes given the lower levels of irrigation in Krasnodar when compared to Edirne. Again, however, the far higher unit values assigned to potato cultivation ensure that this share of total water volume accounts for 99% of the total value of blue water consumed in the high scenario. The total direct use value of blue water consumed in the production of one tonne of potato crisps in the high scenario is estimated at $212, or £163, which is 80% higher than in the low scenario and driven primarily by the larger volume of more highly valued water in potato production.

Table 6.32. Value and volume of blue water used to produce one tonne of potato crisps (high scenario)

Crop (location)	Volume of blue water (m^3)	Unit value (USD 2014)	Value of blue water consumed (USD 2014)	% of total blue water volume	% of total blue water value
Potatoes (East Anglia - UK)	57.16	3.68	210.27	85	99
Sunflower oil (Krasnodar Krai - Russia)	10.17	0.13	1.37	15	1
Total	67.33		211.64	100	100

6.11 Grey water value

As referred to at length in Chapter Three and the previous case studies, grey water refers to the volume of blue water needed to abate pollution. As such, it is assumed here that the unit value of grey water is the same as the unit value of blue water. Ideally, the value of grey water would be equal to the *full range* of in-stream and off-stream values associated with blue water which are no longer available when it is polluted. However, as previously referred to, only the off-stream/extractive values (direct use values) are available to be considered, and thus the applicable unit values are the same as those presented in the previous section. Figures 6.16 and 6.17 below re-state the applicable unit values, and set out the value of grey water along the supply chain based on these unit value estimates, for both the high and low scenarios.

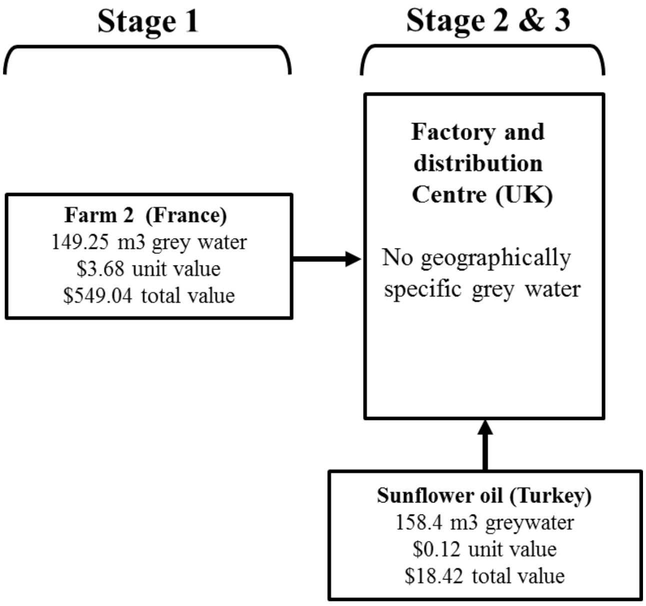

Figure 6.16. Grey water values assigned to each stage of the potato crisp supply chain (low scenario).

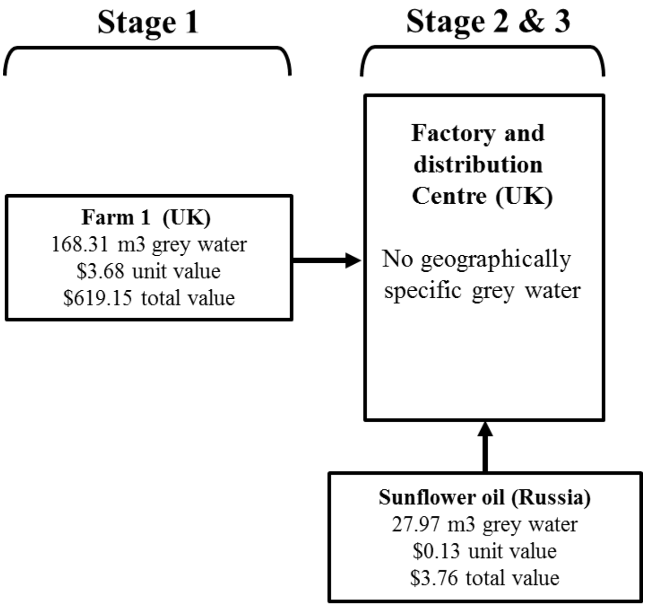

Figure 6.17. Grey water values assigned to each stage of the potato crisp supply chain (high scenario).

Table 6.33 below sets out the total value of grey water in the low scenario. Again, owing to the high unit value assigned to the grey water generated during potato cultivation when compared to sunflower cultivation, the grey water associated with potato irrigation represents 97% of total value even though it accounts for only 49% of the total volume. The total value of grey water in the low scenario is estimated at $567, or approximately £437.

Table 6.33. Value and volume of grey water used to produce one tonne of potato crisps (low scenario)

Crop (location)	Volume of grey water (m^3)	Unit value (USD 2014)	Value of grey water degraded (USD 2014)	% of total grey water volume	% of total grey water value
Potatoes (Nord-Pas-de-Calais - France)	149.25	3.68	549.04	49	97
Sunflower oil (Edirne - Turkey)	158.4	0.12	18.42	51	3
Total	307.65		567.46	100	100

Table 6.34 sets out the total value of grey water in the high scenario. Owing to the far smaller quantity of grey water used in Russia when compared to Turkey, the grey water generated during potato cultivation in East Anglia accounts for a far larger share of total volume (86%) when compared to the volume of irrigation water used in France in the low scenario. However, again because of the high unit value associated with potatoes, the value of the grey water linked with potatoes is disproportionate to its volume, representing as it does, 99% of total grey water value. The total value of grey water in the high scenario is estimated at $623 or £480.

Table 6.34. Value and volume of grey water used to produce one tonne of potato crisps (high scenario)

Crop (location)	Volume of grey water (m^3)	Unit value (USD 2014)	Value of grey water degraded (USD 2014)	% of total grey water volume	% of total grey water value
Potatoes (East Anglia - UK)	168.31	3.68	619.15	86	99
Sunflower oil (Krasnodar Krai - Russia)	27.97	0.13	3.76	14	1
Total	196.28		622.91	100	100

6.12 Green water value

Part Three of Chapter Three set out the approach to valuing green water in light of the available valuation data collected during this study. By way of a recap, green water in this context is not rain water as such but the water that is evapotranspired by the potato crop during its growth phases, or, in other words, it is the volume of water that is usefully used by the crop. As such, it had been anticipated that values for irrigation water *consumed* by the crop would be used as a proxy for the value of green water. However, these were not available in the supply chain locations in stage 1, and as a result, the value of green water will be assumed to be equivalent to the *at source* value of artificially applied irrigation water. In order to estimate at source values, the difference between the mean and median at site and at source values for irrigation water in the USA and ROW value databases, as a whole, was assessed. The largest difference (USA database; mean value) showed that at source values were typically 60% of at site values; the smallest difference (ROW database; median value) showed that at source values were typically 80% of at site values. As a result, these two measures were used to deflate the at site blue water values used above to provide an estimate of the at source value at each stage 1 location. Sensitivity 1 below (or S1) reflects the at source value at 60% of the at site value; sensitivity 2 (or S2) reflects 80%. In many ways this is a crude estimate of the value of green water. However, as mentioned earlier, Aldaya *et al.* (2010a) points to the contemporary significance of green water in the international trade in crops, and thus trying to ensure that the value of green water is incorporated here in some way, is important. What is more, by using a measure of the at source value of water diverted or applied, this is in many ways a conservative estimate of the value of water that is consumed, and thus becomes more defensible.

Figures 6.18 and 6.19 below show the unit values of green water, together with the value of green water consumed at each stage of the supply chain, for both the low and high scenarios.

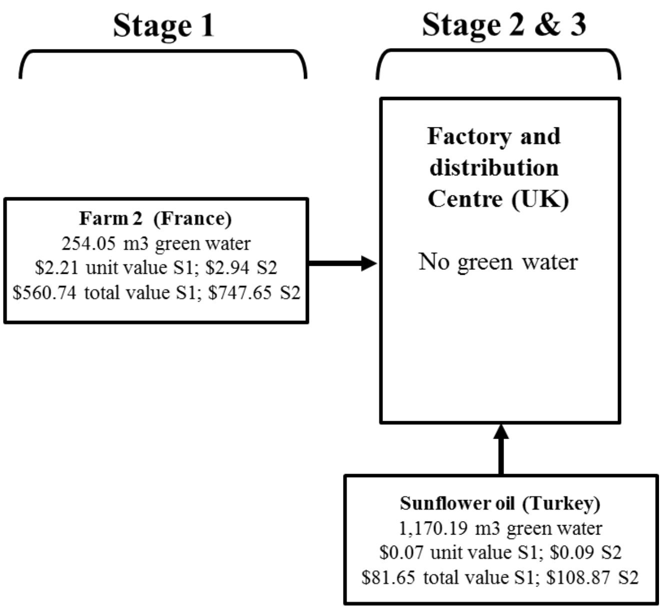

Figure 6.18. Green water values assigned to each stage of the potato crisp supply chain (low scenario).

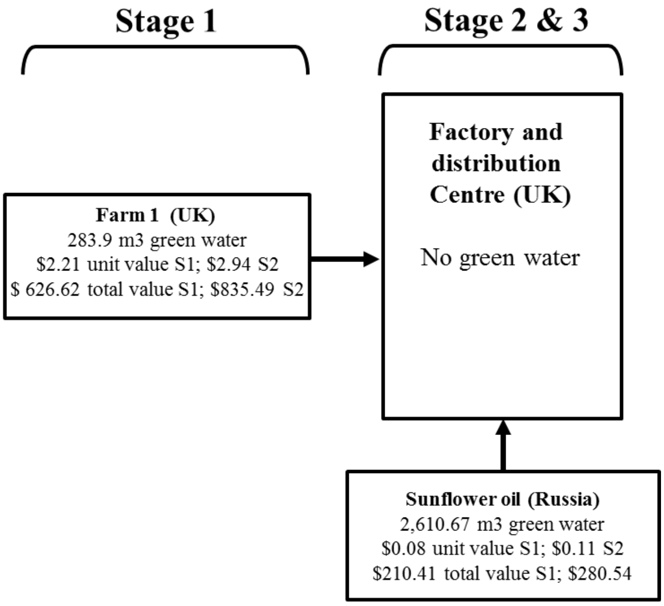

Figure 6.19. Green water values assigned to each stage of the potato crisp supply chain (high scenario).

Tables 6.35 and 6.36 below present the total value of green water in the low and high scenarios. It is evident from these tables that the estimated values for the green water associated with the quantity of potatoes used in one tonne of potato crisps, in both the low and high scenarios, represents a large portion of the *total* value of the potato crop itself. The *Agricultural and Horticultural Development Board* in the UK reported that in June 2017 the wholesale price for a tonne of potatoes, depending on specification, growing location and type, varied between £80 and £500 (AHDB, 2015).[54] Given that one tonne of potato crisps draws on approximately 3.2 tonnes of potatoes (this is based on the product fractions noted in section 6.5.4 and the fact that one tonne of potato crisps contains 69% potato), and that even the lowest green water value estimated in tables 6.35 and 6.36 was $560, or £432 (again using an exchange rate of $1 = 0.77), then assigning a green water value of £135 (£432/3.2) does not stand scrutiny even against even the highest potential crop price. Indeed, a farmer would likely not be willing to pay for green water at these levels, on top of the value of blue water, and ultimately, whether it is a farm crop budget, production function or other method that is used, the value of water in agriculture is tied to the price of the crop. As a result of this analysis, and also that presented previously on green water values in Chapters Four and Five with regard to the wheat and tea crops, the estimated value of green water will be excluded here and the approach to valuing green water will be

[54] Nix (2014) also report 'considerable' variations in the wholesale price of ware potatoes.

revisited in Chapter Seven when the conclusions and recommendations from the project as a whole are set out.

Table 6.35. Value and volume of green water used to produce one tonne of potato crisps (low scenario)

Crop (location)	Volume of green water (m^3)	Unit value S1 (USD 2014)	Unit value S2 (USD 2014)	Value of green water consumed S1 (USD 2014)	Value of green water consumed S2 (USD 2014)	% of total green water volume	% of total green water value
Potatoes (Nord-Pas-de-Calais - France)	254.05	2.21	2.94	560.74	747.65	18	87
Sunflower oil (Edirne - Turkey)	1,170.19	0.07	0.09	81.65	108.87	82	13
Total	1,424.22			642.39	856.52	100	100

Table 6.36. Value and volume of green water used to produce one tonne of potato crisps (high scenario)

Crop (location)	Volume of green water (m^3)	Unit value S1 (USD 2014)	Unit value S2 (USD 2014)	Value of green water consumed S1 (USD 2014)	Value of green water consumed S2 (USD 2014)	% of total green water volume	% of total green water value
Potatoes (East Anglia - UK)	283.90	2.21	2.94	626.62	835.49	10	75
Sunflower oil (Krasnodar Krai - Russia)	2,610.67	0.08	0.11	210.41	280.54	90	25
Total	2,894.57			837.03	1,116.04	100	100

6.13 Implications

In light of the analysis above, tables 6.37 and 6.38 below set out the total value of the blue and grey water used to produce one tonne of potato crisps. As referred to previously, this is only based on those aspects of the supply chain whereby a geographically specific location for water consumption was evident (i.e. it excludes the water used in packaging inputs and the operational and supply chain overhead water footprint components) and it only refers to direct use value. As shown, the total direct use value of the blue and grey water consumed and degraded in the production of one tonne of potato crisps varies between $684 and $835.

Table 6.37. Total value of the blue and grey water used to produce one tonne of potato crisps (low scenario)

Water footprint component	Value USD 2014	Value GBP*
Blue	117	90
Grey	567	437
Total value	684	527

Notes: * 1 USD = 0.77 GBP.

Table 6.38. Total value of the blue and grey water used to produce one tonne of potato crisps (high scenario)

Water footprint component	Value USD 2014	Value GBP*
Blue	212	163
Grey	623	480
Total value	835	643

Notes: * 1 USD = 0.77 GBP.

Tables 6.39 and 6.40 below set out how the total value of blue and grey water breaks down by supply chain stage. In both low and high scenarios, it is the value of the grey (74 or 80%) and blue water (14 or 25%) in potato production that represents the greatest share of total value. However, as mentioned previously, comparisons between the relative value of water in potato and sunflower cultivation should be treated with caution given that water values in potato production are bespoke to that crop, whereas values in sunflower cultivation have been taken from generic low valued crops.

Table 6.39. Total value breakdown by supply chain stage (low scenario)

Stage (location)	Crop	% of total blue and grey value
1 (France - Nord-Pas-de-Calais) Blue water	Potatoes	14
1 (France - Nord-Pas-de-Calais) Grey water	Potatoes	80
1 (Turkey – Edirne) Blue water	Sunflower oil	3
1 (Turkey – Edirne) Grey water	Sunflower oil	3
Total		100

Table 6.40. Total value breakdown by supply chain stage (high scenario)

Stage (location) and water category	Crop	% of total blue and grey value
1 (East Anglia – UK) Blue water	Potatoes	25
1 (East Anglia – UK) Grey water	Potatoes	74
1 (Russia – Krasnodar Krai) Blue water	Sunflower oil	<1
1 (Russia – Krasnodar Krai) Grey water	Sunflower oil	<1
Total		100

Based on the analysis so far, it is clear that the total value associated with the low scenario is approximately 80% of the value of the high scenario. Given that these values are no longer in evidence when water is consumed or degraded, they effectively represent costs, and thus as modelled, sourcing from the combination of countries in the low scenario is preferable to the combination in the high scenario. However, breaking this down further, it is also clear that the value of water consumed and degraded in potato cultivation in France is less than the value of water consumed and degraded in the UK (tables 6.31 to 6.34). This is not surprising given the smaller volumes of blue and grey water used in potato cultivation in France, and the identical unit value that was applied in both locations. Conversely though, for sunflowers, it is evident that the value of blue and grey water is less in Russia than Turkey (tables 6.31 to 6.34). Therefore, if we ignore the large volumes of green water that are present in Russia because, as mentioned, there is not an adequate means of estimating its value, then Russia would be the optimum sunflower oil sourcing location. This is in spite of the higher unit value that has been associated with sunflower oil production in Russia ($0.13 m^3) when compared to Turkey ($0.12 m^3). This overall conclusion – that the optimum combination of sourcing locations would encompass France and Russia – accords with the volumetric water footprint assessment regarding France, and the water stress based conclusions regarding Russia, that were discussed in Part A section 6.9. As such, the analysis here can be viewed as an additional point of reference regarding the impact of the water use along the potato crisp supply chain.

6.14 Sensitivity analysis

Unlike the two previous case studies where the primary ingredients (tea and wheat) were sourced from multiple locations and blended together in the end product, in the two scenarios here (low and high) it is assumed that potatoes are *all* sourced from the UK *or* France and that *all* sunflower oil is sourced either from Turkey *or* Russia. Consequently, it is not necessary to specifically analyse the value of a common quantity of potatoes or sunflower oil because this is identical in both scenarios already modelled. However, as mentioned previously, two sensitivities will be attempted below in order to reflect:

1) How sensitive the conclusions drawn above are to changes in the unit values of water used in the cultivation of sunflower oil in Russia and Turkey, and
2) How sensitive conclusions are to changes in the unit value of water in potato cultivation in France given that it was not possible to estimate a unit value for France which was separate to that applied in the UK.

Sensitivity 1

Sensitivity 1 can be looked at in two ways. The first is by how much would the unit value of water in Turkey need to increase to be equivalent with the value in Russia. There is only a small difference (8%) between the two values ($0.13 - $0.12 = $0.01). The second is by how much would the unit value of water have to *increase* in Russia in order for the *total* value of the blue and grey water used in each location to be comparable. Table 6.41 below shows the total value of the water used to produce one tonne of sunflower oil. As shown the difference in total value is \$101.28 (\$115.25 – 13.97). In order for the total value of the 104 cubic metres of blue and grey water used in Russia to be comparable with the 991 cubic metres used in Turkey (i.e. for it to increase by \$101.28), then the unit value of blue and grey water in Russia would need to *increase* by \$0.97 (or 725%).

Table 6.41. Value and volume of blue and grey water associated with the production of one tonne of sunflower oil in each location

Stage 1 location	Blue water (m^3)	Grey water (m^3)	Total blue and grey water (m^3)	Blue and grey water unit value (USD 2014)	Total value (USD 2014)
Russia (Krasnodar Krai)	27	77	104	0.13	13.97
Turkey (Edirne)	555	436	991	0.12	115.25

This increase in unit value can also be interpreted another way: what the in-stream value (waste assimilation, wildlife habitat, and recreation) of blue and grey water would need to be in Russia in order to call into question the conclusion that total values are lower when compared to Turkey, *ceteris paribus*. When looked at this way, the necessary unit value increase of \$0.97 can be compared to the in-stream value scale which was described in Chapter Three (Part Three). Repeated in Table 6.42 below, this scale shows the minimum, median, and maximum total in-stream values (waste assimilation, wildlife habitat, and recreation) which were observed in the USA (the only country which recorded these values) assuming that all three in-stream ESS are evident in one place.

Table 6.42. In-stream value scale (USD 2014 per m3)

Low	→	Median	→	High
\$ 0.0006		\$ 0.06		\$ 0.6

The income data in Table 6.43 below has been used to adjust the scale in Table 6.42 for relative incomes in Russia using the formula set out by Czajkowski and Scasny (2010) which assumes an income elasticity of one:

$$WTP_{ps} = WTP_{ss}\left(\frac{\overline{I_{ps}}}{\overline{I_{ss}}}\right)\varepsilon$$

where WTP_{ss} is willingness to pay at the study site, WTP_{ps} is the willingness to pay estimate transferred to the policy site, and I_{ss} and I_{ps} are mean income levels at the study and policy sites. ϵ represents the income elasticity of willingness to pay between the mean income levels at the study and policy sites (Czajkowski and Scasny, 2010). These adjusted values are shown in Table 6.44 below.

Table 6.43. Relative income levels in France and Turkey

Country	GNI Per Capita [a]	% of USA GNI Per Capita
USA	52,308.38	100
France	36,628.78	70
Russian Federation	22,616.58	43

Notes: [a] Data sourced from UNDP (2014).

Table 6.44. In-stream value scale Russia (USD 2014 per m3)

Low	→	Median	→	High
0.0003		0.03		0.26

As shown, the increase in blue and grey water unit values which would be needed in Russia in order to produce an equivalent total value to that in Turkey would be far in excess of the highest equivalent in-stream values observed in the USA, which itself assumes that all three in-stream ESS are apparent and valued to the highest extent possible.[55] Given that the USA values are for very water scarce areas of the country, it would seem reasonable to conclude that it is unlikely to be preferable to source sunflower oil in Turkey when compared to Russia, particularly given that there will be additional in-stream values applicable in Turkey which, held constant in this analysis, would widen the value gulf between the two still further.

Sensitivity 2

Table 6.45 below shows the total value of the water used to produce one tonne of potatoes in the UK and France. As with sensitivity 1 above, sensitivity 2 address how much the unit value of blue and grey water would need to *increase* in France in order for the *total* value of the blue, and grey water used in each location to be comparable. As shown, the difference in total value between France and the UK is \$58.86 (\$261.18 – \$202.33). In order for the total value of the 55 cubic metres of blue and grey water used in France to be comparable with the 71 cubic metres used in the UK (i.e. for it to increase by \$58.86), then the unit value of blue and grey water in France would need to increase by \$1.07 (or 29%).

[55] As noted in Chapter Three, in-stream ESS values are additional to agricultural values which are net of extraction costs (i.e. the agricultural value is at source). However, given that at source agricultural values were not available here, the in-stream value scale is applied to at site agricultural values on the assumption of minimal/similar extraction costs across stage 1 sourcing locations.

Table 6.45. Value and volume of blue and grey water associated with the production of one tonne of potatoes in each location

Stage 1 location	Blue water (m^3)	Grey water (m^3)	Total blue and grey water (m^3)	Blue and grey water unit value (USD 2014)	Total value (USD 2014)
Farm 1 – East Anglia	18	53	71	3.68	202.33
Farm 2 - Nord-Pas-de-Calais	8	47	55	3.68	261.18

Again, the necessary increase in unit values can also be interpreted as what the instream value of blue and grey water would need to be in France in order to equalise the total value of the water used to produce a tonne of potatoes with the UK, *ceteris paribus*. Table 6.46 below shows the adjusted in-stream value scale referred to in sensitivity 1. As can be seen, the values associated with in-stream ESS impacted by blue and grey water in France that would be necessary to equalise the total value associated with producing one tonne of potatoes ($1.07) are again far in excess of the equivalent highest in-stream value combination observed in the USA. As a result, if the unit value of water in France is indeed the same as, or similar to, the UK as assumed here, then considerations of in-stream values are unlikely to alter the conclusion that sourcing from France consumes the lowest value of water.

Table 6.46. In-stream value scale France (USD 2014 per m3)

Low		Median		High
0.0004	→	0.04	→	0.43

6.15 Blue water withdrawal value

As well as valuing the water that is consumed and thus no longer available at a place and point in time, this case study is also able to estimate the value of the volume of water withdrawn along the supply chain. Section 6.4 set out the volume of water used during the factory stages (2 and 3), together with an estimate of the volume of water used to cultivate the annual tonnage of potatoes used in the factory (stage 1). The unit value shown in Table 6.29 previously for water use in potato production ($3.68 m^3) is for water *application* and so can be utilised directly to estimate the value of the water applied during stage 1. In addition, Table 6.47 below sets out the estimates of the value of water use in food production. These are for raw intake water and well as water of sufficient quality that it can be used to process food.

Table 6.47. Industrial values used in blue water withdrawal analysis

Source	Method	Value type	Water measure	Original value m^3 (currency)	2014 USD per m^3
Bruneau (2007)	Alternative cost	AV	Process water	0.303 (CAD)	0.36
Bruneau (2007)	Alternative cost	AV	Intake water	0.272 (CAD)	0.32
Renzetti and Dupont (2002)	Cost function	MV	Intake water	0.017 (CAD)	0.02

Based on the unit values in Table 6.29 and 6.47, Table 6.48 below sets out the total value of the water withdrawn or applied in the supply chain. For stage 2, only the 16,500 m^3 used in the brush washers has been included here (see section 6.4) because it is only this portion of the total volume of water used during stages 2 and 3 which appears to be directly related to the processing of the end product. As shown, it is estimated that total blue water withdrawals may support approximately $4 million of value along the supply chain (approximately $3.5 per m^3), the majority during stage 1. Unlike the values that have been assigned to water consumption previously, which were effectively costs, the value of water withdrawn is perhaps not immediately applicable here as there is not a comparator production process to compare the values to. Nonetheless, the total value of the water involved in the annual production of potato crisps, which excludes the value of the water withdrawn in sunflower oil and salt production, provides a significant insight regarding the importance of water to the company.

Table 6.48. Water withdrawal values along the supply chain

Stage (location)	Process	Volume (m^3)	Unit value (USD 2014)	Total value (USD 2014)
1 (UK)	Crop cultivation	1,106,350	3.68	4,069,868
2 (UK)	Brush washers	16,500	0.02 - 0.36	357 - 5,867

6.16 Conclusion

In conclusion, based on the volumes of blue and green water consumed, and grey water degraded, Part A suggested that the supply chain water footprint accounts for approximately 99% of the water burden associated with the potato crisp supply chain. Of this, the vast majority is associated with potatoes and, in particular, sunflower oil, the optimal sourcing locations for which, from a volume perspective alone, were considered to be France and Turkey, respectively. However, Part A also introduced the concept of blue water stress and an analysis of the vulnerability of the water systems along the supply chain. This contradicted the conclusions arrived at when considering volume alone, suggesting as it did, that Edirne and, to a lesser extent, Nord-Pas-de-Calais, represent potential blue water hotspots. In Part B, the value of that portion of the potato crisp water footprint which was specific enough to be subjected to monetary valuation was estimated as varying between $684 and $835 per tonne of potato crisps depending on the scenario. However, this excluded the value of green water which, it was shown, cannot realistically be considered here to be equivalent to the value of artificially applied irrigation water, a conclusion which will be explored further in Chapter Seven.

Beyond this, owing to the granularity of available data on irrigation water values in potato production and, in particular, the absence of a specific value for France, unsurprisingly, the analysis of values in potato production confirmed the conclusion drawn from a volumetric perspective that France was the optimum sourcing location. This conclusion was found to hold unless values in France were at least approximately 29% greater than those in the UK. In addition, it was shown that, based on the same prevailing unit water value as in the UK, substantial enough in-stream ESS values were unlikely to be present in France to change the conclusion that it represents the optimum sourcing location for potatoes from a water perspective. This conclusion however, contradicts the analysis of blue water stress which suggest that France may be a marginal hotspot. With regard to sunflower cultivation, the analysis in Part B contradicted the conclusion in Part A that Turkey represents the optimum sourcing location. However, this conclusion ignores the large volumes of green water used in Russia which it was not possible to assign a monetary value to. Crucially, the implication of this conclusion is that it would be preferable to source sunflower oil from Russia with its predominantly rain-fed conditions, rather than Turkey, which uses substantial quantities of blue and grey water. This conclusion is in accordance with the larger opportunity cost associated with blue water which is noted in the literature. Nevertheless, all the conclusions reached here should be subject to further investigation if decision relevant values were required.

Having now introduced and analysed the three case studies that are the principal subject of this thesis, we now turn to the overall conclusions, recommendations and implications that stem from the project in Chapter Seven.

7. Conclusions, implications and recommendations

Having applied the method that was developed during the course of Chapter Three in the case studies presented in Chapters Four to Six, this chapter now turns to the overall conclusions that stem from the thesis (section 7.1). As part of this, the policy implications that flow from the research (RQ 4) will also be directly addressed here (section 7.2), as will the recommendations for a future research agenda that would better enable the valuation of virtual water flows (section 7.3).

7.1 Conclusions

In this section, the overall conclusions that have come out of the research project are detailed, by RQ, with the exception of RQ4 which is covered separately in section 7.2.

In overview, and as discussed in detail in the following sub-sections, the working aim of the research project which was set out in Chapter One and which is repeated directly below, has largely been achieved here. A new method has been developed and tested in the context of three realistic agri-food case studies which measured the economic, if not societal, value of virtual water. Moreover, this has been used to inform how intra-supply chain water usage might be managed more efficiently.

<u>Aim</u>
To assess the feasibility of, and means to achieve, the measurement of the economic and societal value of virtual water, expressed in monetary terms, within selected global supply chain case studies. Moreover, to explore how this may improve the efficiency of intra-supply chain water usage.

7.1.1 Research question one

The first RQ, set out in section 2.4, focused on whether the existing body of valuation literature can support an approach to monetising virtual water use in a global agri-food supply chain. The review of the unit value literature in Part Two of Chapter Three, which was based on the ESS framework set out in Part One, concluded that there were three main issues with the valuation material that was compiled in the course of this project:

1. A lack of values, in some cases for whole categories of ESS in the framework,
2. The values that were available were skewed in favour of the USA (in particular the South West region of the USA) and contained significant variation in terms of their characteristics even within the same category, and

3. There is a lack of understanding about the application of some value types.

Each of these issues will be tackled in turn below.

Lack of values

As set out in Part Two of Chapter Three, the detailed review of the literature did not find any passive use (non-use values) values, nor did it find any hydrological values which fall within the overarching category of indirect use values. Moreover, the other indirect use values that were part of the ESS framework, namely waste assimilation, wildlife habitat and recreation, returned either an insufficient number of values (wildlife habitat and waste assimilation) to be able to transfer a bespoke value to multiple geographies, or, there was a lack of understanding regarding how this might be achieved (recreation) which will be covered in more detail below. Therefore, a central conclusion here must be that, in light of a large number of significant gaps in the valuation literature, the aim of measuring the full economic <u>and</u> societal value of virtual water has not been possible; only the direct use value of water (i.e. the economic value) appears feasible at the present time.

However, even here there were crucial limitations in terms of the values that were available. Most notably, it was argued that there are only four sources which provide robust estimates of the value of water in industry using appropriate methods. Given that the value of water in industry is principally driven by the use it is put to (i.e. mining, food production etc.), and the fact that the locations where industrial water was used in the case studies were all advanced economies, then it was argued that at least three of the sources provide reasonable estimates of water in this context. Certainly, if the case study supply chains had encompassed multiple industrial stages, in multiple geographies, then it would not have been possible to estimate differentiated values for each site, particularly if the sites were located in a developing country. As it was, the focus on agricultural supply chains meant that each case study, by design, only had one principal industrial location. Therefore, it was the differences in values at the multiple agricultural sites in each case study which were the focus here and which really highlighted the benefit of a monetary approach and the trade-offs that it enables.

Whilst agriculture provided the largest number of valuation estimates of those gathered, it was not possible to generate a pooled model using regression analysis to predict irrigation values in multiple locations that was based on appropriate theoretical foundations.

Therefore, the method that has been used here is limited to supply chains where the agricultural stage is located in geographies for which there are existing irrigation value estimates, or neighbouring geographies which are similar to those for which an estimate is available. This outcome arose because of data paucity and a lack of advancement in the discipline of environmental valuation. Indeed, as will be discussed in section 7.3 which deals with the recommendations that stem from the research, whatever the ESS or use of water, valuation in purely unit value terms appears to be largely overlooked in academic research at present.

Variation within value categories

The USA, and particularly the South West region, accounted for most of the value estimates in each of the ESS categories for which values were available. As a result, the method developed here is perhaps most easily applied to supply chains for which this geography is the principal focus. Indeed, concentrating just on irrigation value estimates which are the key category in the context of agricultural supply chains, outside the USA these are spread very thinly to the extent that in some cases (e.g. Mexico in in the pasta case study India in the tea case study) it was not possible to estimate a value for a specific sub-national region. Indeed, in these cases, an average of other values in the country had to be relied upon. This is certainly not ideal given that irrigation values vary by time and space but represented the best estimate that was available.

However, beyond this geographical focus, the value estimates also contain a number of variances which impact their application. This can be seen across all value categories but it is most readily apparent with irrigation water values which were the focus of the supply chains assessed. Here, values can vary across all the categories set out in Table 3.6. An example of the impact of this variation has been the lack of values for water consumed during irrigation, and thus the need to rely on the value of water withdrawn or applied as a lower bound estimate. Similarly, given the variations in the valuation methods used, some estimates were average values, some were marginal values, and for some the distinction was not clear. Whilst marginal values are the ones which economic theory demands for efficient allocation and thus are the most useful, often this is not possible and the impact that this has on any comparisons of value between locations needs to be considered where possible. In addition, irrigation values were not always available for the specific crop under analysis, most notably tea in Chapter Five. Whilst proxies can be used instead, it is complexities like this that, in effect, further reduce the coverage of the value estimates that have been gathered. The final important area where the irrigation values vary is by the year that they were estimated. Therefore, the estimates may not capture seasonal variances and variances through time, factors which are particularly impact irrigation water values.

Given all this complexity, the method requires that individual value estimates for irrigation water, which are set out in Appendix Four and Appendix Six, are carefully selected by the user, particularly where there is going to be a comparison of water values between locations which is what the monetary valuation approach really enables. Indeed, the estimates in Appendix Four and Appendix Six really need to be examined in detail by the user to ensure that any comparisons of irrigation water values in multiple locations are based on a common scenario, as here. Moreover, even where a common scenario is possible, given that the value of irrigation water is highly variable in both time and space, the value estimates here represent the best estimate of water value in an area, but they are not the value of water in that area. As such, the method described here must be considered, as intended, as one which can provide an initial overview of values in different functional uses (i.e. agriculture, industry and municipal), and to a limited degree, in different geographical areas. Indeed, as mentioned throughout, should there be any requirement for decision relevant values, then these values would need to be investigated further using fully consistent primary valuation techniques.

Lack of knowledge about some value types

There were two principal areas whereby a lack of knowledge about value types inhibited their application. The first of these is with regard to recreation values. Unit values in this category are generated by variations in the level of water flow and, as it stands, a regression analysis of how variations in flow impacts value across studies has not occurred. Indeed, this was not pursued here because at the level of spatiotemporal detail that is our focus, flow variations are not reported in the estimation of water footprints. More importantly though, at the present time there is no framework or guide to approximate how recreational values, on average, decay with distance from the recreation site. Therefore, in the absence of this, recreational values were not included in the values assigned to virtual water. The second area is with regard to green water. As demonstrated, using the value of artificially applied irrigation water as a proxy for green water did not produce a realistic value for the latter. This will be picked up again in the recommendations that follow, but the implication is that the analysis here has focused solely on blue and grey water.

In summary then, the method that has been suggested here can provide an estimation of the value of blue and grey virtual water in a supply chain. However, the value in question would be direct use value only; based on the evidence here it is not possible to estimate societal values (including environmental values) that fall within the categories of indirect and passive uses for the multiple geographies that a global supply chain might span. In addition, the agricultural stages of the supply chain in question would need to span geographies for which appropriate and comparable value estimates exist, and encompass industrial water users which sit within one of the industries covered by the four papers on industrial values referred to earlier. Where this is feasible, the values arrived at should be considered indicative and subject to additional investigation if they were decision relevant. Nonetheless, the method set out in Part Three of Chapter Three estimated the direct use value of the blue and grey virtual water associated with the three case study supply chains as shown in Table 7.1 below. The implications of these monetary values will now be discussed in light of RQs two and three.

Table 7.1 Volume and value of water associated with one tonne of each case study product

Per tonne of product	Tea	Pasta (Low)	Pasta (High)	Potato crisps (Low)	Potato crisps (High)
Total volume green, blue and grey water m^3	6,406	1,658	1,947	2,145	3,344
Monetary value 2014 USD (blue and grey water only)	392	167	174	684	835

7.1.2 Research question two

The second RQ focused on how the value of virtual water is distributed, both by supply chain stage, and geography. As referred to above, the method arrived at enabled the estimation of the direct use value of blue water, and with it grey water, but omitted the value of any green water consumption. However, with these limitations in mind, the principal finding with reference to RQ2 is most evident in the pasta and tea case studies which included a consumer use phase. Indeed, whereas previous water footprint studies – including those presented in the three case studies here – have shown that in agri-food supply chains the *volume* of water consumed and degraded is heavily concentrated in agricultural production of the raw material crop, from a value perspective it is the water used by the end consumer that accounts for the largest share of economic value. For example, in the pasta case study the water consumed when cooking pasta accounted for *circa* 50% of the value of the water in the supply chain, and in the tea case study, the value of the water used to make the tea accounted for approximately 92%.

The concentration of value in the consumer use phase is reflective of the fact that we have used at site values in the case study supply chains in order to be consistent with the value types used in the other stages of each supply chain. As such, the *at site* value reflects the quality of water that is required by the municipal user which is far higher than that required, certainly at the agricultural stage, and likely also the industrial stage as well. Indeed, the values assigned to municipal use in the tea and pasta case studies are derived from market price data, which reflect the price per cubic metre for both water provision *and* waste water services, for the highest block tariff. The inclusion of waste water is in line with the approach adopted by Moran and Dann (2008) when calculating the value of municipal water using the household demand curve method that was also utilised here. In addition, the use of prices reflecting the highest block tariff is also in accordance with the household demand curve method which is based on a reduction from *total* annual usage, which in both the tea and pasta examples, fell within the highest block tariff usage levels. Nonetheless, the basis upon which the municipal or residential water value has been calculated should be borne in mind, as should the fact that the municipal price data that was used did not include any mention of whether the price of the water included charges by the water company for any environmental or social purposes. If these are part of the price, however small, then the resulting value that is calculated may in effect represent more than just the direct use value of municipal water and include a measure of indirect and even passive use value as well, depending on precisely what environmental and social charges were incorporated into the price.

The realisation that the value of water used by the end consumer represents the largest share of the total value of water use in the supply chain, would not lead to a reallocation of intra-supply chain water usage. Moreover, unlike the water used in agriculture, it makes no sense to suggest that some geographies where water is consumed by the end consumer of a product should be prioritised over others. However, the main implication here is perhaps that by attaching a monetary value to the different functions of water in a supply chain, it highlights further the relative importance of water use by the consumer which otherwise may be overlooked in favour of the water use in production.

Aside from the relative value of water used by the end consumer, the case studies have also once again highlighted the limited role that water consumption and degradation can play in the industrial stages of an agri-food product supply chain. In the case of potato crisps, there is no blue water consumption, or grey water degradation, associated with the direct operations of the business. Similarly, the water used in the direct operations of the pasta and tea producers was dwarfed by that used in agriculture. As a result, despite the larger unit values associated with industrial water use, these were not large enough in the tea and pasta case studies to counterbalance the far larger volumes of lower valued water used in the agricultural stages of both supply chains, and thus give industrial water use a prominence it does not enjoy in volume terms. Nonetheless, the value of water in industry could still be an important factor, particularly if: 1) the value of water could be accurately estimated for separate production facilities so as to ascertain the trade-offs associated with reallocation between facilities, or 2) if there was a direct link between less or more water use at the agricultural stage and an impact on water use at the industrial stage. Both of these points will be covered in more detail below.

7.1.3 Research question three

The third RQ in this thesis focused on what the valuation of virtual water flows can reveal about the efficient use and allocation of water in supply chains. In order to structure the conclusions in this context, the results from the method that has been developed and applied here will be discussed around three principal themes: 1) use of the values associated with virtual water as an indicator of impact or risk, 2) use of values as a facilitator of allocative efficiency, and 3) use of values to stimulate productive efficiency. In broad terms, the argument will be advanced here that valuing virtual water can prove relevant to all three of these themes. However, it is the promotion of productive and allocative efficiency that provides the greatest rationale for the method as it is, and certainly as it could evolve to be, and one that may counterbalance any perceived deficiencies in terms of its current ability to provide an indication of impact.

Values as an indicator of impact

The values presented in each of the case study chapters reflect the intensity of willingness to pay (or willingness to accept) for the water use in question. As such they are indicative of the scarcity, or otherwise, of water for that purpose in the location considered. Given that each supply chain case study included multiple crop cultivation locations, the utility of the monetary approach in suggesting the *relative* impact or risk of water use is best illustrated for the agricultural stages of each supply chain (i.e. stage 1) as none of the case studies included multiple locations at subsequent stages. Table 7.2 below sets out the least favourable sourcing locations identified for stage 1 in each of the three case studies based on the monetary approach, and contrasts this with the least favourable locations identified using the volumetric, or water stress, perspectives. As shown, the monetary approach can be used to identify a least favourable location based on unit values alone, or based on volume adjusted values. Similarly, considerations of volume can take account of the blend of the raw material in the end product as in the case of the tea and pasta supply chains, or can be based on a like for like comparison in which the same quantity of raw material is grown at each site. The most useful indicators – volume adjusted values, scarcity and volume (like for like) – are presented first.

Table 7.2 Least favourable sourcing location by approach for each case study

Least favourable sourcing location at stage 1 according to approach	Tea	Pasta	Potato crisp
Volume (like for like)	Indonesia	Montana	UK, Russia
Scarcity	India	Italy, Mexico, SW USA	Turkey, France
Monetary value (volume adjusted)	India	Mexico	Turkey, UK
Volume (blend)	Kenya	Italy	N/A
Monetary value (unit value)	Kenya	France	Turkey *

Notes: * Unit values for UK and France were identical so it is not possible to choose one over the other on this basis.

As indicated in Table 7.2, the monetary approach based on volume adjusted values, concurred with considerations of water scarcity, but contradicted the analysis based on volumes alone (like for like) in the tea case study. Similarly, in the pasta case study, the monetary approach again contradicted the volumetric analysis, identifying (Mexico) as the least favourable sourcing location which was also identified as one area of concern when taking account of water stress. In the potato crisp case study, the conclusion is less clear cut because the number of countries involved was more limited. Nonetheless, the monetary approach again contradicted the volumetric analysis by suggesting that Turkey was the least optimum sunflower oil sourcing location. However, it also contradicted the water stress analysis by suggesting that the UK, rather than France, was the least optimum sourcing location.

Overall, it is quite apparent, particularly in the tea and pasta case studies, that considering the value of the irrigation water, as opposed to just volumes, would lead to a different conclusion regarding sourcing location. In addition, it has also been shown that the least optimal sourcing location identified from a monetary perspective in each case study has been in alignment with considerations of water stress, at least to some degree. As to whether the monetary approach is superior to a water stress perspective as a means of potentially weighting water volumes (as with volume adjusted values), water stress data is both more readily available than monetary values, and importantly, it is often more up to date thus more likely capturing any temporal shifts in water availability in an area. In addition, existing monetary values require interpretation (as indicated), or need to be estimated by economists afresh if decision relevant values are required. However, as will be elaborated on immediately below, monetary values are better understood by businesses than complex LCA outputs based on stress weighted water volumes, are relevant to existing decision making frameworks, and directly enable trade-offs with other inputs in the production process including both financial and natural capital. Therefore, the relative merits of the approaches moving forwards will depend largely on the extent to which the natural capital approach, and the valuation of water within this, is assimilated by businesses. If it is adopted widely, if more valuation material is generated, and if the approach becomes better understood, then valuation of virtual water appears to offer a more useful approach to risk analysis within a supply chain than the present alternatives. However, until that point, the approach identified here is perhaps best considered as an adjunct to current perspectives which may yield additional considerations in an assessment of the impact of virtual water flows.

Values as a means of promoting allocative efficiency

When we talk about allocative efficiency in the context of a supply chain this is not referring to the *same drop* of water flowing to the highest valued use within a geographically delimited area such as a river basin as per the conventional understanding. Indeed, because the supply chain often introduces a large degree of geographical dislocation to this water use, allocative efficiency in this context refers to a broader concept i.e. the decisions that can be made on the basis of having an idea of the relative impact of water use mentioned above. Here, as we have seen, it is possible to quantify the relative values or costs associated with water use in a supply chain and thus quantify the benefits or efficiency gains of souring from one location or another. For example, in the pasta case study it was shown that the cost saving in terms of the blue water consumed and grey water degraded amounted to $148.33 if a tonne of durum wheat was sourced from Orleans (lowest valued area) versus Sonora (highest valued area) *ceteris paribus* (Table 4.20 $149.96 - $1.63). Likewise, the cost saving was $56.50 if a tonne of tea was sourced from Kenya (Nyeri) when compared to India, again *ceteris paribus* (Table 5.15 $77.06 - $20.56). In addition, whilst not fully explored in this thesis, having values such as these would also enable a comparison with other, perhaps financial, costs associated with acting on such signals, as well as non-financial costs, for instance, relating to other environmental impacts (e.g. carbon emissions).

Values as a means of promoting productive efficiency

Productive efficiency here refers to producing more output with less input. Whilst it is not possible to determine such efficiencies based on a snapshot analysis of the three supply chains, by putting a value on the water consumed and degraded in a supply chain which is easily understood in monetary terms, this surely incentivises a reduced water burden within the supply chain. What is more, it is also possible to compare, between supply chains, metrics such as cubic metres per dollar as shown in Table 7.3 below. Whilst this would not necessarily lead to a pasta producer trying to emulate a tea producer, it way well incentivises one individual tea or pasta producer to try and emulate another tea or pasta producer which is demonstrating best practice, assuming perfect competition and symmetry of information. Similarly, whilst only the potato crisp supply chain had volumes of water withdrawn, as opposed to consumed, available to it, the value of this metric again might be one that could be compared between supply chains.

Table 7.3 Cubic metres per dollar

Per tonne of product	Tea	Pasta (Low)	Pasta (High)	Potato crisps (Low)	Potato crisps (High)
Cubic metres per 2014 USD	16.34	9.93	11.19	3.14	4.00

In addition, as will be discussed in the next section which looks at RQ4, particularly when values such as these are internalised, factors such as relative exchange rate fluctuations, and even variations in tax regimes become relevant in the resource optimism decision. Indeed, it is important geographically variable factors such as these which volumetric analysis, or considerations of water stress, cannot take into account and which are only enabled by a monetary focus.

Finally, whilst the valuation of green water has not been possible in this context, if the recommendations set out in section 7.3 are acted on, this would also enable a consideration of the trade-offs between green and blue water consumption at each stage of a supply chain that are suggested by commentary of their relative opportunity costs (see Turner *et al.*, 2004, p.37).

7.2 Implications

In this section, we deal directly with RQ 4 which focused on how regulatory instruments might be designed in response to the valuation of virtual water flows. The discussion here draws on section 2.2 which introduced the relevant theory and concepts.

Whilst the valuation exercise in this thesis has looked at many different water values, which fall in several different categories, in this context it is helpful to divide the discussion into the implications of the valuation of blue water, and the implications of the value of grey water.

Starting with the latter, it is quite clear that grey water, which as we have seen is the volume of water necessary to assimilate pollutants, represents an external *social cost* which is illustrated by the divergence between MPC and MSC in Figure 2.2 in Chapter Two. As such, having estimates of the value of grey water, such as those presented in the case studies and repeated in Table 7.4 below, provides an indication of, for example, the size of any pollution tax that might be imposed. For instance, in the tea case study, the value of grey water was shown to be approximately $23 per tonne of tea. Therefore, imposing a pollution tax of this magnitude, which would be split between the farmers in the regions where tea is grown as these were the principal sources of grey water in the supply chain, would theoretically internalise the externality. Alternatively, any pollution tax could be levied on the end producer of the product, or ultimately the end consumer, rather than the farmer, for the pollution associated with the entire supply chain. However, the method utilised here to value grey water, as explained at length earlier, has been to treat blue water consumption and grey water degradation the same i.e. to assume that the value of grey water is the direct use value it could have been put to if it had not been polluted. Whilst this is fine for a methodology such as this which is used for initial assessment, if a more precise figure was required for a regulatory exercise, then the cost of abating the pollution itself, rather than the opportunity cost of the dilution water, might be more appropriate.

Table 7.4 Volume and value of grey water only associated with one tonne of each case study product

Per tonne of product	Tea	Pasta (Low)	Pasta (High)	Potato crisps (Low)	Potato crisps (High)
Total volume grey water m^3	143	207	237	308	196
Monetary value 2014 USD	23	62	69	567	623

Notes: For each of the three products grey water is associated with the agricultural production stage only. The disparity between potato crisp (high) and (low) arises because the majority of extra grey water in the low scenario is of low value. However, there is additional higher valued grey water in the high scenario which counteracts this.

The alternative to imposing a tax to disincentivise grey water (the Polluter Pays Principle discussed in section 2.2) is to incentivise farmers to adopt different methods (the Provider Gets Principle) that do not produce grey water. In this scenario, farmers could be compensated for not irrigating their crops up to the value of the blue water used in irrigation, the idea being that in the absence of irrigation there would be less run-off from fertilisers and therefore less grey water. However, this is a rather blunt instrument given that the famer is being compensated for the value of irrigation water use, and not strictly the value of the fertilisers, even though the former, in conjunction with the latter, gives rise to grey water.

Moving on to the value of blue water specifically, as discussed at length in Chapter Three, the initial aim here had been to treat this as equal to the value of the full range of instream <u>and</u> off-stream ESS that are impacted by its consumption. However, following the review of the literature in Part Two of Chapter Three it became apparent that it was not possible to provide geographically specific estimates of the *instream* value categories i.e. waste assimilation, wildlife habitat, recreation, hydrological and passive use. Had these estimates been available however, then the value of the instream ESS impacted by blue water consumption could, like grey water, have been treated as a societal cost to be internalised using the concepts set out in section 2.2. However, the direct use (off-stream) value of water should not be treated in the same way. The easiest way of illustrating why is to consider the value of irrigation water (the focus of the analysis in the agri-food supply chain case studies) estimated by a farm crop budget (the most common method used to value irrigation water). As Gibbons (1987, p.29) states, the residual value which is attributed to water using this method is estimated by subtracting non-water input costs from crop revenue and thus represents the '…maximum amount the farmer could pay for water and still cover costs of production.' As such, using a means to internalise this value does not make sense and may simply ensure that the crop was not grown. Indeed, for this reason, water which is an intermediate input into production and whose value is subject to a derived demand (i.e. in this case from the value of the crop), should be treated with caution and separate to the value of water pollution or instream ESS consumed when discussing the internalisation of virtual water values.

7.3 Recommendations

In this section, we set out several recommendations that would better enable the valuation of virtual water flows. These are grouped around the following five sub-headings which are covered in turn: 1) unit values as a focus of academic research, 2) the value of water as an intermediate input into production, 3) water flow and the link to value, 4) general recommendations for the environmental valuation literature, and 5) the valuation of green water.

Unit values as a focus of academic research

What has perhaps become most apparent during the course of this research project is the fact that in the water valuation literature, unit values are somewhat of a poor relation, particularly in academia. This situation has arisen because the focus of much academic research in the field of water valuation appears to be on incrementally improving the SP and RP techniques that are used to value water for recreational purposes, which is commonly reported in denominations other than volume such as the value per day of the recreational activity. Indeed, the value of water as an intermediate input into production (i.e. water used in agriculture and industry), which is the water category that most lends itself to unit valuation and has been the focus here, is as Young and Loomis (2014) recognise, a relatively neglected area in academic research. The principal exception to this is the valuation work in the South West region of the USA which, owing to the pressure on water resources in that area, together with the established nature of environmental valuation in policy and practice, has engendered the vast majority of the unit valuation literature that was available to this study. This work has been conducted, in large part, as a means of improving inter-sectoral allocation of water resources. Whether it be for this rationale though, or to aid the valuation of virtual water flows, the first recommendation here is that the field of environmental valuation reconsiders the units that it currently reports values in, and where possible, at least supplements current practices with a consideration of volumetric water value. In the context of a recreational value study, this would involve taking account of water flow in the study region, as for example Loomis and McTernan

(2014) have done. Outside of this, and as addressed at greater length below, the second recommendation must be that the academic literature also looks to diversify its focus away from recreational values to include some of the other value categories looked at in this context, principally the value of water as an intermediate input to production. Furthermore, to do so in unit value terms because for businesses which understand their water burden in volumetric terms, this would obviously have the greatest application.

The value of water as an intermediate input into production – agriculture and industry

The principal challenge for the academic and specialist literature in the field of industrial water values would appear to be the development of a reliable, but easily applicable, method for estimating unit values in this context. Unlike the value of water in agriculture, which can be estimated using the residual method (farm crop budget) which can be calculated with a simple spreadsheet, industrial water does not have an equivalent procedure. Indeed, the residual method has no purchase for the calculation of industrial values because water is often such a small input into production. As a result, as we have seen, the value of water in industry is a relatively unstudied area, consigned to four studies which were deemed to use rigorous, albeit complex, methods which have consequently not seen wide adoption. Whilst the value of water in industry is principally driven by the use it is put to, if more values were available, then a better understanding of how value varies by industrial facility might be possible, which would in turn allow the dynamic reallocation of water between facilities, and also potentially between stages in a supply chain.

In terms of agricultural values, as mentioned there is a simple and proven technique for estimating unit values in this context. Therefore, the suggestion here is that there should be a degree of standardisation as to components in a crop budget, and that this standardised budget should be deployed as widely as possible for a set of standard crops during different stages of the growing season. Whilst the values that a farm crop budget arrives are not ideal from the point of view of economic theory – they provide average rather than marginal values – they nonetheless would provide a common yardstick as to relative values in different areas and at different times which could lead to the dynamic reallocation of water resources in response to, for example, emerging climate patterns. Agricultural research stations already compile much of the necessary information to make this approach a reality and the relative simplicity of the method would aid its uptake. The challenge would come in terms of adopting a standardised approach but certainly in countries such as the USA, with common reporting standards, this would seem possible. Indeed, had more values been available for the USA, then the predictive regression model that was attempted in Chapter Three might have been feasible, which would in turn may have enabled the transfer of values outside the USA.

Water flow and the link to value

The discussion of recreational values in Part Three of Chapter Three introduced the idea that, correctly conceived, recreational value is a function of water flow. Indeed, the same can also be said for other in-stream ecosystem goods and services. For example, Alvarez *et al.* (2016) suggest that assimilation of pollutants is aided by increased levels of flow. However, whilst we have seen that there are a limited number of recreational value studies that have been conducted based on variations in river flow (Table 3.30), as Emerton (2005) suggest in the context of the Pangani River Basin, the link between ESS values and water flow within a basin is an unexplored area. Therefore, an increased emphasis on this relationship in the academic literature is the next recommendation here. In order to do this, as mentioned in Chapter Three and recognised by Alvarez *et al.* (2016), the different river profiles (i.e. width and depth) would need to be taken into account owing to the fact that 10,000 cubic feet per second may be high flow in one river but low flow in another. None of this is to say that unit values for some ESS are not possible without this understanding. Indeed, as we have seen, there are water values available for wildlife habitat that are the product of water market transactions i.e. water has been leased or purchased in a specific volume for a specific wildlife purpose. However, additional research into the fundamental relationship between river flow and the value of ecosystem goods and services in a basin would provide a more fundamental and comprehensive understanding.

General recommendations for the environmental valuation literature

It was a common occurrence, when analysing the studies that provided unit values for use in this project, for key pieces of information which would have aided understanding, and crucially, their use in a benefits transfer exercise, to be unclear. Therefore, Table 7.5 below sets out a suggested set of parameters that each study could usefully report in tabulated short form on page one. In addition, drawing on the omissions that impacted the regression analysis of agricultural values in Part Three of Chapter Three, Table 7.6 sets out the parameters that should be reported for agricultural values. Whilst many of these might seem obvious, and certainly a trained environmental economist could infer many of them based on the methodology used, if the values are to be of use to as wide an audience as possible, then key parameters should be made as explicit as possible.

Table 7.5. Suggested parameters to be clearly reported in valuation studies

Parameter	Explanation
Water category/ESS	e.g. Waste assimilation, agriculture etc.
Valuation method	The approach that has been utilised in the valuation exercise.
Valuation year	Year that values refer to as distinct from year of publication.
Econometric focus	WTA/WTP/other
Theoretical underpinnings	Demand curve or non-demand curve.
Welfare measure	e.g. Marshallian or Hicksian welfare measure.
Value type	Average or marginal value
Volumetric measure	Cubic metre or acre feet
Location of study	Geographic location and spatial scale of valuation exercise

Table 7.6. Suggested parameters to be clearly reported in irrigation water value studies

Parameter	Explanation
Crop type	Description of crop type and sub-type.
Approximate crop value	Value of crop in USD per tonne.
Spatial scale	Regional or field level.
Irrigation water price	Price of irrigation water if any.
Farmers adjustment options	Changes in acreage, crop mix, irrigation schedule and technology.
At site/at source	If using residual method, have water delivery costs been subtracted?
Time frame	Long or short run.
Water use type	Diversion, application or consumption.

It is entirely possible for the discipline of environmental valuation to coalesce around a set of best practice guidelines such as these. Indeed, as evidenced by, for example, the adoption of common standards for WFA, stress weighted water footprints (ISO 14046), and even carbon footprinting, other disciplines have evolved as such and managed to adopt a standardised methodological approach.

The valuation of green water

The method for valuing green water that was set out in Part One of Chapter Three aimed to treat the value of rain water evapotranspiration as equal to the value of artificially applied irrigation water that is evapotranspired i.e. consumed. Given the lack of values for water consumption, Part Three of Chapter Three suggested that values for water withdrawal or application would be utilised to value green water instead, thus providing greater conservatism as these represent lower bound estimates of the value of water consumption. However, it became clear in the case studies (particularly pasta and potato crisps) that, given the large volumes of green water involved, this was producing value estimates which were unrealistic. Therefore, the next recommendation here is that the value of green water is a focus of academic research moving forward in order to give what Aldaya *et al.* (2010a) describe as a ‘strategic resource’ a voice in economic analyses.

The issues involved in doing so are best illustrated using the example of a simple farm crop budget. In the situation where a crop is irrigated naturally (i.e. by rainfall) and artificially, two farm crop budgets would ideally be conducted. The first would focus on the crop under rain fed conditions, and the second under artificial irrigation, with the uplift in value of the crop less any *additional* non-water costs being the residual at site value attributable to the irrigation water. However, in this scenario a residual value could be attributed to the rain fall using the first farm crop budget i.e. the value of the crop prior to artificial irrigation less non-water costs. Of the irrigation water studies cited in Chapter Three, one – Bakker *et al.* (1999) – did take account of rain fall in their residual analysis. However, they did so in the context of a single crop budget that simply divided the residual value from an artificially irrigated crop by the total evapotranspiration whether through rainfall or artificial irrigation. The problem here though is that this does not split out the value of rain from the value of artificial irrigation, and as we have seen, the value of the latter does not appear to be synonymous with the former. Indeed, it may be that artificial irrigation, which is likely applied later in crop production, adds disproportionate value when compared to rain fall which is evapotranspired earlier in the crop growth cycle, depending on the climatic conditions. It is issues such as this that the environmental valuation literature still needs to tackle.

Having now discussed the conclusions that have emanated from the research – as well as the broader implications and recommendations – the final chapter now moves on to address how these can be reconciled with the relevant literature base, and how the research might be extended in the future.

8. Synthesis and reflections

In this chapter, the conclusions that have resulted from the research project are synthesised with the empirical literature on virtual water, as well as the relevant body of (welfare economic) theory (section 8.1). Following this, section 8.2 will reflect on potential future research scope.

8.1 Synthesis

This section begins by discussing the contribution that the research has made to welfare economic theory, before addressing how it adds to the empirical literature on the measurement and assessment of virtual water flows.

Theoretical context

The principal theoretical contribution of this thesis has not necessarily been the refinement of existing welfare economic theory, but rather, the application and interpretation of this theory in a novel context (i.e. within a supply chain setting). What this has shown is that whilst the theory of pareto optimality, as traditionally applied in a specific water basin, would suggest that the *same drop* of water should flow to the *highest* valued use (for example from low valued agricultural uses to higher values industrial uses), when the backdrop is a geographically disparate supply chain, then value needs a different interpretation. Indeed, in the context of the supply chain, it is differences in the relative value of *different drops* of water that become the focus and how these values, or more precisely the loss of these values when water is consumed or degraded, provide an indication of impact and thus have the potential to inform trade-offs between locations. Here, *lowest* relative value become the focus, as described in the three case study chapters, and efficiency is judged in terms of the minimisation of the loss of value from water consumption and degradation.

In addition, Chapter Seven also argued that the mechanisms for internalising externalities that were set out in Chapter Two and which flow from welfare economic theory, apply to some categories of water use but not others. Principally, it was suggested that water pollution, and the consumption of in-stream ESS, could be viewed as social costs appropriate for remedy and internalisation. However, it was also suggested that where water is used as an intermediate input into production – in agricultural and industrial settings – that this was not appropriate to internalise given the current methods that are used to value these water uses. Indeed, as far as the author is aware, the theoretical literature has not dealt with how the value of water as an intermediate input, once estimated, should be internalised.

Empirical context and methodological contribution to the assessment of virtual water flows

One of the principal contributions of this thesis has been the thorough review of the unit value literature that was conducted in Chapter Three (Part Two). What this, and the analysis in Chapter Three (Part Three) has shown, is that the approaches identified in Chapter Two that emanate from the grey literature and which claim to be able to place a unit value on some/many of the in-stream or off-stream water related ESS, and to be able do so for any potential geography, should be treated with a degree of caution based on the evidence assembled here. It is quite conceivable that the literature search conducted in Chapter Two (Part Two), focused on the specialist environmental valuation databases, did not capture every possible unit value estimate that has been published. Indeed, the paper by Frederick *et al.* (1996) that appears to the basis of the approach adopted by Trucost referenced a number of sources that were no longer available and thus which could not form part of this review. Nonetheless, for categories such as waste assimilation, there is such a dearth of evidence available – Frederick *et al.* (1996) only referenced one paper themselves – that it seems questionable as to whether a robust and rigorous means of generating a bespoke and geographically specific value for any potential geography could be possible without the addition of a large number of additional studies. Indeed, a central contribution of this thesis is, I feel, that it highlights just how much unit valuation of water has been neglected by the discipline of environmental valuation in favour of other areas of focus.

The grey literature aside, the method that was developed here – and certainly the improved method that it may stimulate – has direct relevance to the literature on the empirical measurement and assessment of virtual water flows. Indeed, as agued throughout, a monetary approach to virtual water potentially offers an easily comprehensible metric that is accessible to a variety of audiences in a similar manner to the original concept of the water footprint which has become a pervasive advocacy tool that has been widely applied. Whilst there is no doubt that the estimation of water values requires considerable prior knowledge, certainly if original value studies are to be conducted, it offers a superior metric when gauging impact when compared to the complex outputs generated by LCA and the concept of stress weighting water footprints. In addition, it has the potential to dovetail with existing business decision making frameworks and it offers the means by which the broader goals of WFA in terms of ensuring that water rich areas assume the burden of producing products which are water intensive, can be realised. This is not to say that economics can solve all ills; certainly, there may be environmental and social goals which go beyond those that are captured by TEV and the focus on anthropocentric values which is inherent in welfare economics. As such this method should be considered as an adjunct to WFA and one which has the potential to enable some of the aims that are embodied in the *Water Footprint Assessment Manual* regarding sustainability assessment (phase 3 of WFA).

Finally, the principle of monetising the impacts of supply chains is beginning to be realised in the wider supply chain management literature. We have already noted the study by Pizzol *et al.* (2015), and O'Rourke (2014) have also described attempts to monetise supply chain impacts as 'the most ambitious' approach to assessing sustainability in a supply chain. This study is therefore also of direct relevance in this more general context as well.

8.2 Self reflections

One of the principal discoveries here has been the gaps and limitations associated with the water unit valuation literature to the extent that, on reflection, I would likely explore other options further before repeating the research described if I were doing it again. Indeed, had I known the limitations associated with the value base compiled, and the substantial amount of time that it would take just to review the body of environmental valuation literature in order to arrive at this, then a more limited study which made use of primary valuation techniques may have been preferable. Such an approach is developed in the future research agenda below. However, having said this, I am also mindful that the approach described in the following section is largely the product of the familiarity with the literature – and the myriad of techniques that it contains – that I gained from having to go through so much of it in order to compile the value base used here. Therefore, it is only having done the research described that I am in a position to set this out; it is certainly not something that I was aware of at the inception of this research project.

This reflection aside, I hope that the method developed here acts as a catalyst to the environmental valuation discipline both in terms of enabling an improved method for valuing virtual water to be developed, but also in terms of giving greater emphasis to business relevant metrics. At the present time it feels as if practice has to make do with the data generated in an academic context which is sometimes divorced from business application. The best example is the ongoing focus in the academic water literature on recreational values rather than the value of water as an intermediate input into production, measured in unit value terms, which would appear to have more immediate application.

8.3. Future research agenda

It has been very interesting to get to this point and discover, after a thorough review of the literature, the data gaps in, and limitations of, the existing environmental valuation literature base. If I were to take the research project further, then the five key gaps identified in section 7.3 are those that the discipline needs to address in the long term. However, of these five areas, the one that I could tackle immediately would be to conduct an original valuation study of water use in agriculture using the farm crop budget technique and wider scenario briefly referred to in section 7.3.

The aim here would be to deploy a simple and easily replicable primary technique to discover how values vary, at higher levels of spatiotemporal resolution, during the agriculture stages of a supply chain, which as we have seen, is the stage most susceptible to valuation and the one which is most responsible for water consumption and degradation. Ultimately this is a departure from the level of spatiotemporal detail that was selected here. However, until such time as additional unit values for the various water related ESS become available – and that is the hope following this project – then a primary study such as this may further reinforce the *principle* of monetary valuation of virtual water and thus engender the additional value estimates required.

The steps involved in the original valuation study would include:

1. Decide on a common definition of the constituent components of the farm crop budget.
2. Identify a large geographical area which encompasses a wide variety of climatic conditions and levels of water stress, and one for which the common items in a farm crop budget are recorded in a consistent format. Given the prevalence of the environmental valuation in the USA, this would seem like an ideal setting.
3. Select a common set of crops and an intra-season reporting pattern.
4. Monitor the average values that are generated and how these vary seasonally, by geography and by crop type.

The merit of this would be that fully consistent water values would be generated in multiple different areas, facing different background conditions, which would be of direct relevance to a policy decision. Indeed, if the farm crop budget locations were designed to coincide with the specific areas where a product's raw materials were sourced from (for example, the wheat sourcing locations in the pasta case study) then this would be of direct relevance to producing company. Moreover, such an approach could also collect additional data, for example, on carbon emissions, which would enable an understanding of the trade-offs associated with different water and non-water cost and benefits.

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Appendix 1. List of sources – USA database

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1	Amirfathi, P., Narayanan, R. and Bishop, B. and Larson, D.	A methodology for estimating instream flow values for recreation.	Utah Water Research Laboratory.	1985	Report	N	4	
2	Aylward, B., Seely, H., Hartwell, R. and Dengel, J.	The economic value of water for agricultural, domestic and industrial uses: A global compilation of economic studies and market prices.	Ecosystem Economics. Report prepared for UN FAO.	2010	Report	N	1,3,5	
3	Bernardo, D.J., Whittlesey, N.K., Saxton, K.E. and Bassett, D.L.	Valuing irrigation water: A simulation/mathematical programming approach.	Water Resources Bulletin, Vol. 24, No.1.	1988	Journal	N	1	
4	Bishop, R., Boyle, K., Welsh, M., Baumgartner, R. and Rathbun, P.	Glen canyon dam releases and downstream recreation: An analysis of user preferences and economic values.	Madison, WI: Heberlein-Baumgartner Research Services.	1987	Book	Y	4	Brown, T.C. (1991) Water for wilderness areas: Instream flow needs, protection and economic value.
5	Bollman, F.H.	A simple comparison of values: Salmon and low value irrigation crops.	Paper for the Association of California Water Agencies.	1979	Working paper	N	5	Gibbons, D.C. (1987) The economic value of water.

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Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
6	Booker, J.F. and Colby, B.G.	Competing water uses in the Southwestern United States: Valuing drought damages.	Water Resources Bulletin, Vol. 31, No.5.	1995	Journal	N	1,3,4	
7	Brewer, J., Glennon, R. Ker, A. and Libecap, G.	Water markets in the west: prices, trading and contractual reforms.	NBER Working paper 13002.	2007	Working paper	N	1,3	
8	Brown Jr, G.M. and McGuire C.H.	A socially optimal pricing policy for a public water agency.	Water Resources Research, Vol. 3., no. 1.	1967	Journal	Y	1	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
9	Brown, T.C., Harding, B.L. and Payton, E.A.	Marginal economic value of streamflow: A case study for the Colorado River Basin.	Water Resources Research , Vol. 26, No. 12.	1990	Journal	N	1	
10	Bush, A.	Is the Trinity River dying?	Instream Flow Needs, Vol. 2 (American Fisheries Society).	1976	Journal	N	5	Gibbons, D.C. (1987) The economic value of water.
11	Bush, D.B. and Martin, W.E.	Potential costs and benefits to Arizona agriculture of the Central Arizona Project.	University of Arizona College of Agriculture Technical Bulletin No. 254.	1986	Report	N	1	
12	Bustic, V. and Netrusil, N.R.	Valuing water rights in Douglas County, Oregon, using the hedonic price method.	Journal of American Water Resources Association, Vol. 43 No. 3.	2007	Journal	N	1	

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13	Butcher, W.R., Whittlesey, N.K. and Osborn, J.F.	Economic values of water in a systems context.	Report prepared for the National Water Commission. Report #NWC-SBS-72-048.	1972	Report	Y	1	Turner, K., Georgiou, S., Clark, R., Brouwer, R. and Burke, J. (2004). Economic value of water resources in agriculture: From the sectoral to a functional perspective on natural resources management.
14	Chang, C. and Griffin, R.C.	Water marketing as a reallocative institution in Texas.	Water Resources Research Vol. 28., No.3.	1992	Journal	N	1,3	
15	Colby, B.G.	The economic value of instream flows: Can instream values compete in the market for water rights?	L.J. Macdonnell, T.A. Rice, and S.J. Shupe. (eds). Instream flow protection in the west. Natural Resources Law Centre. University of Colorado School of Law. Boulder, Colorado.	1989	Book	Y	4	Postel, S. and Carpenter, S. (1997). Freshwater Ecosystem Services and Moore, D. and Willey, Z. (1991) Water in the American West.
16	Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M.	A model for estimating demand for irrigation water on the Texas High Plains.	Texas Water Resources Institute Technical Report No. 68 Texas A & M University.	1975	Report	Y	1	Gibbons, D.C. (1987). The economic value of water.

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17	Cooper, J. and Loomis, J.B.	Testing whether waterfowl hunting benefits increase with greater water deliveries to wetlands.	Environmental and Resource Economics, Vol. 3, No. 6.	1993	Journal	N	1,4	
18	Creel, M. and Loomis, J.	Recreation value of water to wetlands in the San Joaquin Valley: Linked multinomial logit and count data trip frequency model.	Water Resources Research Vol. 28., No. 10.	1992	Journal	N	4	
19	D'Arge, R.C.	Quantitative water resources basin planning: An analysis of the Pecos river basin, New Mexico.	Water Resources Research Institute in cooperation with the Department of Economics. University of New Mexico.	1970	Report	Y	2	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
20	Daubert, J.T. and Young, R.A.	Recreational demand for maintaining instream flows: A contingent valuation approach.	American Journal of Agricultural Economics, Vol. 63, No. 4.	1981	Journal	N	1,4	
21	Duabert, J.T. and Young, R.A.	Economic benefits from instream flow in a Colorado mountain stream.	Colorado Water Resources Research Institute, Completion Report No. 91 (Study Number 36).	1979	Report	N	4	

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22	Duffield, J.W., Neher, C.J. and Brown, T.C.	Recreation benefits of instream flow: Application to Montana's Big Hole and Bitterroot rivers.	Water Resources Research, Vol. 28, No. 9.	1992	Journal	N	1,4	
23	Fadali E. and Shaw, W.D.	Can recreation values for a lake constitute a market for banked agricultural water?	Contemporary Economic Policy, Vol. 16, No.4.	1998	Journal	N	1	
24	Faux, J. and Perry, G.M.	Estimating the irrigation water value using hedonic price analysis: A case study in Malheur County, Oregon.	Land Economics, Vol. 75, No. 3.	1999	Journal	N	1	
25	Gayle, S., Willitt, S.H. and Robertson, C.E.	The economic value of water used to irrigate field crops in central and southern Arizona.	Department of Agricultural Economics Report No. 9. Tuscon: University of Arizona.	1975	Report	Y	1	Gibbons, D.C. (1987). The economic value of water.
26	Gibbons, D.C.	The economic value of water.	A study for Resources for the Future.	1987	Book	N	1,3,4,6	
27	Gisser, M., Lansford, R.R., Gorman, W.D., Creel, B.J. and Evans B.	Water trade-off between electric energy and agriculture in the Four Corners Area.	Water Resources Research , Vol. 15, No. 3.	1979	Journal	N	1	

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28	Gray, S.L. and Young, R.A.	The economic value of water for waste dilution: Regional forecasts to 1980.	Journal of the Water Pollution Control Federation, Vol. 46. No, 7.	1974	Journal	N	6	
29	Hansen, L.T. and Hallam, A.	National estimates of the recreational value of streamflow.	Water Resources Research, Vol. 27, No. 2.	1991	Journal	N	1,4	
30	Harpman, D.A.	The value of instream flow used to produce a recreational fishery.	Doctoral Dissertation. Fort Collins: Colorado State University (Department of Agricultural and Resource Economics).	1990	Thesis/dissertation	Y	4	Brown, T.C. (1991). Water for Wilderness areas: Instream flow needs, protection and economic value.
31	Hartman, L.M. and Anderson, R.L.	Estimating the value of irrigation water from farm sales data in north eastern Colorado.	Journal of Farm Economics, Vol. 44, No. 1.	1962	Journal	N	1	
32	Hartman, L.M. and Anderson, R.L.	Estimating irrigation water values.	Technical bulletin 81, Colorado Agricultural Experiment Station.	1963	Report	Y	1	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
33	Hastay, M.	The Colombia River as a resource: Socioeconomic consideration of diversion and the value of Colombia river water, part A.	State of Washington Water Research Centre.	1971	Report	Y	6	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.

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34	Houk, E. E., Frasier, M. and Taylor R.G.	Evaluating water transfers from agriculture for reducing critical habitat water shortages in the Platte Basin.	Journal of Water Resources Planning and Management, Vol. 133, No. 4.	2007	Journal	N	1	
35	Howe, C.W. and Ahrens, W.A.	Water resources of the Upper Colorado River Basin: Problems and policy alternatives.	In El-Ashry (Ed), M.T. and Gibbons, D.C. (eds). (1988). Water and the Arid Lands of the Western United States: A World Resources Institute book.	1988	Book	N	1	
36	Howe, C.W. and Easter, K.W.	Interbasin transfers of water: Economic issues and impacts.	Johns Hopkins Press.	1971	Book	Y	1	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
37	Johnson, N.S. and Adams, R.M.	Benefits of increased streamflow: The case of the John Day River steelhead fishery.	Water Resoures Research, Vol. 24, No.11.	1988	Journal	N	4	
38	Kane, J. and Osantowski, R.	An evaluation for water reuse using advanced waste treatment at a meat packing plant.	Proceedings of the 35th Industrial Wastes Conference.	1981	Conference paper	Y	2	Gibbons, D.C. (1987) The economic value of water.
39	Kelso, M.M., Martin W.E. and Mack L.E.	Water supplies and economic growth in an arid environment.	University of Arizona Press Study Number 18.	1974	Report	Y	1	Gibbons, D.C. (1987) The economic value of water.

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
40	Kneese, A.V.	Water resources development and uses.	Kansas City Federal Reserve Bank.	1959	Report	Y	2	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
41	Kneese, A.V.	Appendix to Wollman et al. The Value of water in alternative uses.		1962	Report	Y	2	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
42	Kollar, K.L., Brewer, R. and McAuley, P.H.	An analysis of the price/cost sensitivity of water use in selected manufacturing industries.	Bureau of Domestic Commerce staff study (Water Resources Council).	1976	Report	N	2	
43	Lacewell, R.D., Sprott, J.M. and Beattie, B.R.	Value of irrigation water with alternative input prices product prices and yield levels.	Texas Water Resources Institute Technical Report No. 58 Texas A & M University.	1974	Report	Y	1	Gibbons, D.C. (1987) The economic value of water.
44	Lansford Jr, N.H. and Jones J.L.	Recreational and aesthetic value of water using hedonic price analysis.	Journal of Agricultural and Resource Economics Vol. 20, No. 2.	1995	Journal	N	4	

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
45	Lofting, E.M. and McGaughey, P.H.	Economic evaluation of water part 3: An interdisciplinary analysis of the California water economy.	Sanitary Engineering Research Laboratory. College of Engineering and School of Public Health. University of California Berkley. Contribution no. 67, Water Resources Centre.	1963	Report	Y	2	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
46	Loomis, J. and McTernan, J.	Economic value of instream flow for non-commercial white-water boating using recreation demand and contingent valuation methods.	Environmental Management, Vol. 53, No. 3.	2014	Journal	N	4	
47	Loomis, J.B.	Comparing households' total economic values and recreation value of instream flow in an urban river.	Journal of Environmental Economics and Policy, Vol. 1, No. 1.	2012	Journal	N	4	
48	Loomis, J.B. and Creel, M.	Recreation benefits of increased flows in California's San Joaquin and Stanislaus Rivers.	Rivers, Vol. 3, No. 1.	1992	Journal	N	4	

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
49	Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J.	Expanding institutional arrangements for acquiring water for environmental purposes: Transactions evidence for the western United States.	International Journal of Water Resources Development, Vol 19, No.1.	2003	Journal	N	4,5	
50	Meritt, L.B. and Mar, B.W.	Marginal value of dilution water.	Water Resources Research, Vol. 5 No. 6.	1969	Journal	N	6	
51	Moore, D. and Willey, Z.	Water in the American West: Institutional evolution and environmental restoration in the 21st century.	University of Colorado Law Review, Vol. 62.	1991	Journal	N	1,4,5	
52	Naeser, R.B. and Bennett, L.L.	The cost of non-compliance: The economic value of water in the middle Arkansas River Valley.	Natural Resources Journal, Vol. 38, No. 3.	1998	Journal	N	1	
53	Narayanan, R.	Evaluation of recreational benefits of instream flows.	Journal of Leisure Research, Vol. 18, No. 2.	1986	Journal	Y	4	Brown, T.C. (1991) Water for Wilderness areas: Instream flow needs, protection and economic value.
54	Neher, C.J.	The economic value of instream flows in Montana: A travel cost model approach.	Masters dissertation. University of Montana.	1989	Thesis/dissertation	N	4	

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
55	Postel, S. and Carpenter, S.	Freshwater ecosystem services.	Dailey, G. (ED). Natures Services: Societal dependence on natural ecosystems. Washington D.C.: Island Press.	1997	Book	Edited	4,5	
56	Powell, S.T.	Relative economic returns from industrial and agricultural water uses.	American Water Works Association Vol.48, No. 8.	1956	Journal	N	2	
57	Renshaw, E.F.	Value of an acre foot of water.	American Water Works Association, Vol. 50, No.3.	1958	Journal	N	1,2,5,6	
58	Russell, C.S.	Industrial water use.	Section 2, Report to the National Water Commission, Resources for the Future.	1970	Report	Y	6	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
59	Shulstad, R.N., Cross, E.D. and May, R.D.	The estimated value of water in Arkansas.	Arkansas Farm Research, Vol. 27 No.6.	1982	Journal	N	1	
60	Shumway, C.R.	Derived demand for irrigation water: The California Aqueduct.	Southern Journal of Agricultural Economics.	1973	Journal	Y	1	Gibbons, D.C. (1987) The economic value of water.
61	Torell, L.A., Libbin, J.D. and Miller, M.D.	The market value of water in the Ogallala Aquifer.	Land Economics, Vol.66, No.2.	1990	Journal	N	1	

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
62	Walsh, R.G., Auckerman, R. and Milton, R.	Measuring benefits and the economic value of water in recreation on high country reservoirs.	Water Resources Research Institute, Completion Report No. 103. Colorado State University.	1980	Report	N	4	
63	Walsh, R.G., Ericson, R., Arostegy, D. and Hansen, M.	An empirical application of a model for estimating the recreation value of instream flow.	Water Resources Research Institute, Completion Report No.101. Colorado State University.	1980	Report	N	4	
64	Ward, F.	Optimally managing wild and scenic rivers for instream flow benefits.	Proceedings of the national river recreation symposium. Baton Rouge: Louisiana State University.	1985	Proceedings	Y	4	Loomis, J. (1987). The economic value of instream flow: Methodology and benefit estimates for optimum flows.
65	Ward, F.A.	Economics of water allocation to instream uses in a fully appropriated river basin: Evidence from a Mexico wild river.	Water Resources Research, Vol. 23, No. 3.	1987	Journal	N	4	
66	Ward, F.A., Roach, B.A. and Henderson, J.E.	The economic value of water in recreation: Evidence from the California drought.	Water Resources Research, Vol. 32, No. 4.	1996	Journal	N	4	

Appendix 1. List of sources – USA database

Ref #	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
67	Washington State University Agricultural Research Centre	Irrigation development potential and economic impacts related to water use for the Yakima River Basin.		1972	Report	Y	1	Gibbons, D.C. (1987) The economic value of water.
68	Wollman, N. et al.	The value of water in alternative uses.	University of New Mexico Press.	1962	Report	Y	2	Young, R.A. and Gray, S.L. (1973). Economic value of water: Concepts and empirical estimates.
69	Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S.	Economic value of water: Concepts and empirical estimates.	Technical Report to the National Water Commission. NTIS No. PB210356. Springfield, VA., National Technical Information Service.	1973	Report	N	1,2,3	

* 1 = Agriculture, 2= Industry, 3= Municipal, 4= Recreation, 5= Wildlife habitat, 6= Waste Assimilation.

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
70	Ahmad, M., Masih, I. and Turral, H.	Diagnostic analysis of spatial and temporal variation in crop water productivity: A field scale analysis of the rice-wheat cropping system in Punjab.	Journal of Applied Irrigation Science, Vol. 39, No 1.	2004	Journal	N	1	
71	Al-Ghuraiz, Y. and Enshassi, A.	Ability and willingness to pay for water supply service in the Gaza Strip.	Building and Environment, Vol. 40, No.8.	2005	Journal	N	3	
72	Al-Weshah, R.	Optimal use of irrigation water in the Jordan Valley: A case study.	Water Resources Management, Vol. 14, No.5.	2000	Journal	N	1	
73	Anielski, M. and Wilson, S.J.	Counting Canada's natural capital: Assessing the real value of Canada's boreal ecosystems.	Canadian Boreal Initiative and The Pembina Institute.	2005	Report	N	3	
74	Arias Rojo, R. H.	Not available (see secondary reference)	Secretaría de Medio Ambiente y Recursos Naturales Comisión Nacional Forestal Comisión Nacional Forestal Banco Mundial Banco Mundial.	2007	Report	Y	1	EVRI. (2011).

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
75	Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F.	Multiple uses of water in irrigated areas: A case study from Sri Lanka.	International Water Management Institute SWIM paper 8.	1999	Working/ discussion paper	N	1	
76	Banda, B.M., Farolfi, S. and Hassan, R.M.	Estimating water demand for domestic use in rural South Africa in the absence of price information.	Water Policy, Vol.9, No.5.	2007	Journal	N	3	
77	Birol, E., Koundouri, P. and Kountouris, Y.	Assessing the economic viability of alternative water resources in water-scarce regions: Combining economic valuation, cost-benefit analysis and discounting.	Ecological Economics, Vol. 69, No.4.	2010	Journal	N	1	
78	Birol, E., Koundouri, P. and Kountouris, Y.	Farmers' demand for recycled wastewater in Cyprus: A contingent valuation approach.	Wastewater Reuse-Risk Assessment, Decision Making and Environmental Security.	2007	Journal	N	1	
79	Bowen, R. and Young, R.	Financial and economic irrigation net benefit functions for Egypt's Northern Delta.	Water Resources Research, Vol. 21, No 9.	1985	Journal	N	1	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
80	Bruneau, J.	Economic value of water in the South Saskatchewan River Basin.	Martz, L., Bruneau, J. and Rolfe, T. (eds). (2007). Climate Change and Water: SSRB final technical report.	2007	Report	Edited	1,2	
81	Calatrava, L.J. and Sayadi, S.	Economic valuation of water and willingness to pay analysis with respect to tropical fruit production in south-eastern Spain.	Spanish Journal of Agricultural Research, Vol. 3, No.1.	2005	Journal	N	1	
82	El Chami, D., Knox, J.W., Daccache, A. and Weatherhead, E.K.	The economics of irrigating winter wheat in a humid climate: A study in the East of England.	Agricultural Systems, Vol. 133.	2015	Journal	N	1	
83	Emerton, L (ed).	Values and rewards: Counting and capturing ecosystem water services for sustainable development.	IUCN Water, Nature and Economics Technical Paper No. 1, IUCN: The World Conservation Union, Ecosystems and Livelihoods Group Asia.	2005	Report	N	3	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
84	Emerton, L., Erdenesaikhan, N., De Veen, B., Tsogoo, D., Janchivdorj, L., Suvd, P., Enkhtsetseg, B., Gandolgor, G., Dorisuren, C., Sainbayar, D. and Enkhbaatar, A.	The economic value of the upper Tuul ecosystem, Mongolia.	Mongolia Discussion Papers, East Asia and Pacific Sustainable Development Department. Washington, D.C.: World Bank.	2009	Report	N	1,2,3	
85	Esmaeili, A. and Vazirzadeh, S.	Water pricing for agricultural production in the south of Iran.	Water Resource Management, Vol. 23, No.5.	2009	Journal	N	1	
86	Hellegers, P.J.G.J. and Perry, C.J.	Water as an economic good in irrigated agriculture: Theory and practice.	The Hague: Agricultural Economics Research Institute.	2004	Report	N	1	
87	Hernandez-Sancho, F and Sala-Garrido, R.	Technical efficiency and cost analysis in wastewater treatment processes: A DEA approach.	Desalination, Vol. 249, No.1.	2009	Journal	N	6	
88	Hernández-Sancho, F., Molinos-Senante, M. and Sala-Garrido, R.	Economic valuation of environmental benefits from wastewater treatment processes: An empirical approach for Spain.	Science of The Total Environment, Vol. 408, No.4.	2010	Journal	N	6	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
89	Hussain, I., Turral, H., Molden, D. and Ahmad, M.D.	Measuring and enhancing the value of agricultural water in irrigated river basins.	Irrigatin Science, Vol. 25, No.3.	2007	Journal	N	1	
90	Kadigi, R., Mdoe, N., Oshimogo, G. and Moradet, S.	Water for irrigation or hydropower generation? Complex questions regarding water allocation in Tanzania.	Agricultural Water Management, Vol. 95, No.8.	2008	Journal	N	1	
91	Kanyoka, P., Farolfi, S. and Morardet, S.	Households' preferences and willingness to pay for multiple use water services in rural areas of South Africa: An analysis based on choice modelling.	Water SA, Vol. 34, No. 6.	2008	Journal	N	3	
92	Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M.	Determining the economic value of irrigation water in Kerio Valley Basin (Kenya) by residual value method.	Journal of Economics and Sustainable Development, Vol 6, No.7.	2015	Journal	N	1	
93	Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P.	Mapping the financial benefits of sprinkler irrigation and potential financial impact of restrictions on abstraction: A case-study in Anglian Region.	Journal of Environmental Management, Vol.58, No.1.	2000	Journal	N	1	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
94	Kulshreshtha, S.N.	Economic value of groundwater in the Assiniboine Delta aquifer in Manitoba.	Ottawa, Ontario: Environment Canada.	1994	Report	N	1,2,3	
95	Kulshreshtha, S.N. and W.J. Brown.	The economic value of water for irrigation: A historical perspective.	Canadian Water Resources Journal, Vol.15, No. 3.	1990	Journal	N	1	
96	Kumar, S.	Analysing industrial water demand in India: An input distance function approach.	National Institute of Public Finance and Policy Working Paper No 12/2004.	2004	Working/ discussion paper	N	2	
97	Larson, B., Minten, B. and Razafindralambo, R.	Unravelling the linkages between the Millennium Development Goals for poverty, education, access to water and household water use in developing countries: Evidence from Madagascar.	Journal of Development Studies, Vol. 42, No.1.	2006	Journal	N	3	
98	Latinopoulos, P., Tziakas, V. and Mallios, Z.	Valuation of irrigation water by the hedonic pricing method: A case study in Chalkidiki, Greece.	Water, Air and Soil Pollution: Focus, Vol. 4, No.4.	2004	Journal	N	1	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
99	Lavee, D.	Cost-benefit analysis of constructing a filtration plant for the national water carrier in Israel.	Water and Environment Journal, Vol. 23, No.4.	2009	Journal	N	6	
100	Louw, D.B. and van Schalkwyk, H.D.	The true value of irrigation water in the Oilfants river basin: Western Cape.	Agrekon, Vol. 36, No.4.	1997	Journal	N	1	
101	Martinez-Paz, J. M. and Perni, A.	Environmental cost of groundwater: A contingent valuation approach.	International Journal of Environmental Research, Vol. 5, No. 3.	2011	Journal	N	1	
102	McCartney, M. P., Lankford, B. A., Mahoo, H.	Agricultural water management in a water stressed catchment: Lessons from the RIPARWIN project.	Colombo, Sri Lanka: IWMI.	2007	Report	N	3	
103	Menegaki, A. N., Hanley, N. and Tsagarakis, K.P.	The social acceptability and valuation of recycled water in Crete: A study of consumers' and farmers' attitudes.	Ecological Economics, Vol. 62, No.1.	2007	Journal	N	1	
104	Molinos-Senante, M., Hernandez-Sancho, F and Sala-Garrido, R.	Cost-benefit analysis of water-reuse projects for environmental purposes: A case study of Spanish wastewater treatment plants.	Journal of Environmental Management, Vol. 92, No.12.	2011	Journal	N	6	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
105	Molinos-Senante, M., Hernandez-Sancho, F and Sala-Garrido, R.	Economic feasibility study for wastewater treatment: A cost-benefit analysis.	Science of the Total Environment, Vol. 408, No.20.	2010	Journal	N	6	
106	Moran, D. and Dann, S.	The economic value of water use: Implications for implementing the Water Framework Directive in Scotland.	Journal of Environmental Management, Vol. 87, No.3.	2008	Journal	N	1	
107	Muller, R.A.	The value of water in Canada.	Canadian Water Resources Journal, Vol. 10, No.4.	1985	Journal	N	1,3	
108	Nieuwoudt, W.L., Backeberg, G.R. and Du Pleiss, H.M.	The value of water in the South African economy: Some implications.	Agrekon, Vol.43, No. 2.	2004	Journal	N	1,3	
109	Pazvakawambwa, G.T., Van Der Zaag, P.	The value of irrigation water in Nyanyadzi smallholder irrigation scheme, Zimbabwe.	1st WARFSA/ WaterNet Symposium: Sustainable Use of Water Resources, Maputo, 1-2 November 2000.	2001	Working/ discussion paper	N	1	
110	Perez-Pineda, F. and Quintanilla-Armijo, C.	Estimating willingness-to-pay and financial feasibility in small water projects in El Salvador.	Journal of Business Research, Vol. 66, No.10.	2013	Journal	N	3	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
111	Puente Gonzalez, A.	Not available (see secondary reference).	Secretaría de Medio Ambiente y Recursos Naturales Comisión Nacional Forestal Banco Mundial.	2007	Report	Y	1	EVRI. (2011).
112	Qureshi, M. E., Connor, J., Kirby, M. and Mainuddin, M.	Economic assessment of acquiring water for environmental flows in the Murray Basin.	Australian Journal of Agricultural and Resource Economics, Vol. 51, No.3.	2007	Journal	N	1	
113	Raje, D., Dhobe, P. and Deshpande, A.	Consumers willingness to pay for municipal supplied water: A case study.	Ecological Economics, Vol 42, No.3.	2002	Journal	N	3	
114	Renwick, M. E.	Valuing water in a multiple-use system: Irrigated agriculture and reservoir fisheries.	Irrigation and drainage systems, Vol. 15, No.2.	2001	Journal	N	1	
115	Renzetti, S. and Dupont, D.P.	The value of water in manufacturing.	CSERGE Working Paper ECM 03-03.	2002	Working/ discussion paper	N	2	
116	Rodgers, C. and Hellegers, P.J.G.J	Water pricing and valuation in Indonesia: A case study of the Brantas river basin.	EPT Discussion Paper 141. International Food Policy and Research Institute.	2005	Working/ discussion paper	N	1	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
117	Rogers, P., Bhatia, R. and Huber, A.	Water as a social and economic good: How to put the principle into practice.	TAC Background Paper No.2. Global Water Partnership and Swedish International Development Cooperation Agency.	1998	Working/ discussion paper	N	1,2,3	
118	Samarawickrema, A. and Kulshreshtha, S.	Value of irrigation water for crop production in the South of Saskatchewan River Basin.	Canadian Water Resources Journal, Vol. 33, No.3.	2008	Journal	N	1	
119	Tan, R. P. and Bautista, G.M.	Metering and a water permits scheme for groundwater use in Cagayan de Oro.	Economy and Environment Program for Southeast Asia. Research Report No. 2003-RR8. International Development Research Centre.	2003	Report	N	2	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
120	Turpie, J., Day, E., Ross-Gillespie, V. and Louw, A.	Estimation of the water quality amelioration value of wetlands: A case study of the Western Cape, South Africa.	Environment for Development Discussion Paper 10-15, Environment for Development Initiative and Resources for the Future, Washington DC.	2010	Working/ discussion paper	N	6	
121	Walker, I., Ordonez, F., Serrano, P. and Halpern, J.	Pricing, subsidies and the poor: Demand for improved water services in Central America.	The World Bank.	2000	Report	N	3	
122	Wang, H and Lall, S.	Valuing water for Chinese industries: A marginal productivity assessment.	Applied Economics, Vol. 34.	2002	Journal	N	2	
123	Wang, H., Xie, J. and Li, H	Domestic water pricing with household surveys: A study of acceptability and willingness to pay in Chongquing, China.	The World Bank.	2008	Report	N	3	
124	Whittington, D., Lauria, D.T. and Mu, X.	A study of water vending and willingness to pay for water in Onitsha, Nigeria.	World Development, Vol 19, No. 2-3.	1991	Journal	N	3	

Appendix 2. List of sources – Rest of World database

Ref#	Author	Title	Source	Date	Document type	Secondary source Y/N	Category *	Abbreviated secondary reference
125	Yokwe, S.C.B.	Investigation of the economics of water as used by smallholder irrigation farmers in South Africa.	University of Pretoria (Agricultural Economics Extension and Rural Development).	2005	Thesis	N	1	
126	Zetina-Espinosa, A. M., Mora-Flores, J. S., Martínez-Damián,M. A., Cruz-Jiménez, J. and Téllez-Delgado, R	Economic value of water in irrigation district 044, Jilotepec, Estado de Mexico.	Agricultura, Sociedad y Desarrollo 2, Vol. 10. Colegio de Postgraduados. México.	2013	Journal	N	1	
26 **	Gibbons, D.C.	The Economic Value of Water.	A study for Resources for the Future.	1987	Book	N	3	
69**	Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S.	Economic value of water: Concepts and empirical estimates.	Technical Report to the National Water Commission, NTIS no. PB210356 (Springfield, Va., National Technical Information Service, 1972).	1973	Report	N	2	

* 1 = Agriculture, 2= Industry, 3= Municipal, 4= Recreation, 5= Wildlife habitat, 6= Waste Assimilation. ** Common to both USA and ROW databases.

Appendix 3

Bureau of Economic Analysis - Table 1.1.9. Implicit Price Deflators for Gross Domestic Product.

Year	Deflator
2015	109.998
2014	108.828
2013	106.913
2012	105.214
2011	103.311
2010	101.221
2009	100
2008	99.246
2007	97.337
2006	94.814
2005	91.988
2004	89.12
2003	86.735
2002	85.039
2001	83.754
2000	81.887
1999	80.065
1998	78.859
1997	78.012
1996	76.699
1995	75.324
1994	73.785
1993	72.248
1992	70.569
1991	68.996
1990	66.773
1989	64.392
1988	61.982
1987	59.885
1986	58.395
1985	57.24
1984	55.466
1983	53.565
1982	51.53
1981	48.52
1980	44.377
1979	40.706
1978	37.602
1977	35.135
1976	33.083
1975	31.361
1974	28.703
1973	26.337
1972	24.978
1971	23.941
1970	22.784
1969	21.642
1968	20.627
1967	19.786
1966	19.227
1965	18.702
1964	18.366
1963	18.088
1962	17.886
1961	17.669
1960	17.476
1959	17.237
1958	17.001
1957	16.625
1956	16.091
1955	15.559
1954	15.298
1953	15.157
1952	14.972
1951	14.716
1950	13.745
1949	13.581

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (California Central Valley)	At site	Short	Withdrawal	Unknown	Water market transaction	187.93	187.93	N/A	Reported
Bernardo, D.J., Whittlesey, N.K., Saxton, K.E. and Bassett, D.L. (3)	USA (Washington - Columbia River basin)	At site	Short	Application / delivery	Unknown	Other	0 – 215.12	25.70 [a]	Median	Summarised
Bernardo, D.J., Whittlesey, N.K., Saxton, K.E. and Bassett, D.L. (3)	USA (Washington - Columbia River basin)	At site	Short	Consumption	Unknown	Other	0 - 259.16	46.56 [a]	Median	Summarised
Bernardo, D.J., Whittlesey, N.K., Saxton, K.E. and Bassett, D.L. (3)	USA (Washington - Columbia River basin)	At site	Short	Application / delivery	Unknown	Other	132.42 – 217.02	164.97 [b]	Median	Summarised
Booker, J.F. and Colby, B.G. (6)	USA (Western Colorado)	At site	Unknown	Consumption	Unknown	Linear Programming	18.81	18.81	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Colorado Front Range)	At site	Unknown	Consumption	Unknown	Linear Programming	20.66	20.66	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Wyoming)	At site	Unknown	Consumption	Unknown	Linear Programming	19.28	19.28	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Utah)	At site	Unknown	Consumption	Unknown	Linear Programming	19.28	19.28	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (New Mexico)	At site	Unknown	Consumption	Unknown	Linear Programming	18.81	18.81	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (San Juan Chama Export)	At site	Unknown	Consumption	Unknown	Linear Programming	18.81	18.81	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (New Mexico - Nevajo Indian Irrigation Project)	At site	Unknown	Consumption	Unknown	Linear Programming	83.12	83.12	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Arizona - Central Arizona Project)	At site	Unknown	Consumption	Unknown	Linear Programming	41.79	41.79	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Colorado River Indian Tribe)	At site	Unknown	Consumption	Unknown	Linear Programming	22.36	22.36	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Booker, J.F. and Colby, B.G. (6)	USA (Arizona – Yuma)	At site	Unknown	Consumption	Unknown	Linear Programming	30.84	30.84	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (California)	At site	Unknown	Consumption	Unknown	Linear Programming	41.95	41.95	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Arizona)	At site	Short	Withdrawal	Unknown	Water market transaction	60.02	60.02	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (California)	At site	Short	Withdrawal	Unknown	Water market transaction	71.07	71.07	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Colorado)	At site	Short	Withdrawal	Unknown	Water market transaction	24.84	24.84	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Idaho)	At site	Short	Withdrawal	Unknown	Water market transaction	8.05	8.05	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Montana)	At site	Short	Withdrawal	Unknown	Water market transaction	6.34	6.34	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (New Mexico)	At site	Short	Withdrawal	Unknown	Water market transaction	24.88	24.88	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Oregon)	At site	Short	Withdrawal	Unknown	Water market transaction	8.87	8.87	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Texas)	At site	Short	Withdrawal	Unknown	Water market transaction	31.66	31.66	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Utah)	At site	Short	Withdrawal	Unknown	Water market transaction	7.80	7.80	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Washington)	At site	Short	Withdrawal	Unknown	Water market transaction	18.72	18.72	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Wyoming)	At site	Short	Withdrawal	Unknown	Water market transaction	4.33	4.33	N/A	Reported
Brown Jr, G.M. and McGuire C.H. (8)	USA (California – San Joaquin Valley)	Unknown	Long	Application / delivery	Unknown	Other	93.50	93.50	N/A	Reported
Brown, T.C., Harding, B.L. and Payton, E.A. (9)	USA (Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	39.93 – 91.26	65.59 [c]	Median	Summarised
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	High	Other	407.56	407.56	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	High	Other	374.56	374.56	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	High	Other	270.65	270.65	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	High	Other	191.50	191.50	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	Low	Other	186.63	186.63	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Maricopa county)	At site	Short	Application / delivery	Low	Other	123.96	123.96	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pima county)	At site	Short	Application / delivery	High	Other	223.52	223.52	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pima county)	At site	Short	Application / delivery	Low	Other	67.14	67.14	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pima county)	At site	Short	Application / delivery	Low	Other	112.45	112.45	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pinal county)	At site	Short	Application / delivery	High	Other	186.83	186.83	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pinal county)	At site	Short	Application / delivery	Low	Other	128.30	128.30	N/A	Reported
Bush, D. and Martin, W. (11)	USA (Arizona – Pinal county)	At site	Short	Application / delivery	Low	Other	108.27	108.27	N/A	Reported
Bustic, V. and Netrusil, N.R. (12)	USA (Oregon - Douglas County)	In stream	Long	Application / delivery	Unknown	Hedonic	16.96 – 33.91	20.35 [d]	Median	Summarised

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Butcher, W.R., Whittlesey, N.K. and Osborn, J.F. (13)	USA (unknown)	Unknown	Unknown	Unknown	High	Other	155.80	155.80	N/A	Reported
Butcher, W.R., Whittlesey, N.K. and Osborn, J.F. (13)	USA (unknown)	Unknown	Unknown	Unknown	High	Other	14.12	14.12	N/A	Reported
Chan, C. and Griffin, R.C. (14)	USA (Texas – Lower Rio Grande Valley)	At site	Long	Application / delivery	High	Farm crop budget	-8.77 – 193.99	80.12 [e]	Median	Summarised
Chan, C. and Griffin, R.C. (14)	USA (Texas – Lower Rio Grande Valley)	At site	Short	Withdrawal	Unknown	Water market transaction	14.75	14.75	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Short	Application / delivery	Low	Linear Programming	19.62	19.62	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Short	Application / delivery	Low	Linear Programming	95.64	95.64	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Short	Application / delivery	Low	Linear Programming	137.33	137.33	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Short	Application / delivery	High	Linear Programming	169.21	169.21	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Short	Application / delivery	High	Linear Programming	176.57	176.57	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Linear Programming	0.00	0.00	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Linear Programming	31.88	31.88	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Linear Programming	34.33	34.33	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Long	Application / delivery	High	Linear Programming	61.31	61.31	N/A	Reported
Condra, G.D., Lacewell, R.D., Sprott, M. and Adams, M. (16)	USA (Texas – High Plains)	In stream	Long	Application / delivery	High	Linear Programming	58.86	58.86	N/A	Reported
Cooper, J. and Loomis, J.B. (17)	USA (California)	Unknown	Unknown	Unknown	Unknown	Other	24.30	24.30	N/A	Reported
Daubert, J.T. and Young, R.A. (20)	USA (Colorado – Poudre River)	Unknown	Short	Unknown	Unknown	Other	5.06 – 130.53	23.47 [f]	Median	Summarised
Duffield, J.W., Neher, C.J. and Brown, T.C. (22)	USA (Montana – Ravali County)	At site	Long	Application / delivery	Low	Yield comparison	61.69	61.69	N/A	Reported
Duffield, J.W., Neher, C.J. and Brown, T.C. (22)	USA (Montana – Beaverhead County)	At site	Long	Application / delivery	Low	Yield comparison	29.30	29.30	N/A	Reported
Fadali E. and Shaw, W.D. (23)	USA (California)	At site	Short	Withdrawal	High	Water market transaction	197.16	197.16	N/A	Reported
Faux, J. and Perry, G.M. (24)	USA (Oregon - Treasure Valley/Malheur County)	In stream	Long	Application / delivery	Unknown	Hedonic	13 – 63.57	27.45 [g]	Median	Summarised

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	Low	Farm crop budget	-17.17	-17.17	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	Low	Farm crop budget	-14.71	-14.71	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	Low	Farm crop budget	26.98	26.98	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	Low	Farm crop budget	26.98	26.98	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	High	Farm crop budget	93.19	93.19	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	High	Farm crop budget	93.19	93.19	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Maricopa County)	At site	Long	Application / delivery	Low	Farm crop budget	120.17	120.17	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	Low	Farm crop budget	-2.45	-2.45	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	Low	Farm crop budget	29.43	29.43	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	Low	Farm crop budget	61.31	61.31	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	High	Farm crop budget	134.88	134.88	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	High	Farm crop budget	95.64	95.64	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pinal County)	At site	Long	Application / delivery	Low	Farm crop budget	107.90	107.90	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	Low	Farm crop budget	-14.71	-14.71	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	Low	Farm crop budget	12.26	12.26	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	Low	Farm crop budget	36.79	36.79	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	Low	Farm crop budget	61.31	61.31	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	High	Farm crop budget	56.40	56.40	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Pima County)	At site	Long	Application / delivery	High	Farm crop budget	122.62	122.62	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	Low	Farm crop budget	26.98	26.98	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	Low	Farm crop budget	19.62	19.62	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	Low	Farm crop budget	58.86	58.86	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	Low	Farm crop budget	56.40	56.40	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	High	Farm crop budget	80.93	80.93	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	High	Farm crop budget	39.24	39.24	N/A	Reported
Gayle, S., Willitt, S.H. and Robertson, C.E. (25)	USA (Arizona – Cochise County)	At site	Long	Application / delivery	Low	Farm crop budget	164.31	164.31	N/A	Reported
Gibbons, D.C (26)	USA (Idaho)	At site	Unknown	Application / delivery	Low	Production Function	1,711.74	1,711.74	N/A	Reported
Gibbons, D.C (26)	USA (Washington)	At site	Unknown	Application / delivery	Low	Production Function	144.69	144.69	N/A	Reported
Gibbons, D.C (26)	USA (Washington)	At site	Unknown	Application / delivery	Low	Production Function	353.14	353.14	N/A	Reported
Gibbons, D.C (26)	USA (Washington)	At site	Unknown	Application / delivery	Low	Production Function	691.56	691.56	N/A	Reported
Gibbons, D.C (26)	USA (California)	At site	Unknown	Application / delivery	High	Production Function	174.12 – 316.35	245.24 [h]	Median	Summarised
Gibbons, D.C (26)	USA (California)	At site	Unknown	Application / delivery	Low	Production Function	956.42	956.42	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	Low	Production Function	36.79	36.79	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	Low	Production Function	53.95	53.95	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	Low	Production Function	61.31	61.31	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	High	Production Function	137.33	137.33	N/A	Reported
Gibbons, D.C (26)	USA (New Mexico)	At site	Unknown	Application / delivery	Low	Production Function	61.31	61.31	N/A	Reported

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Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Gibbons, D.C (26)	USA (New Mexico)	At site	Unknown	Application / delivery	High	Production Function	149.59	149.59	N/A	Reported
Gibbons, D.C (26)	USA (New Mexico)	At site	Unknown	Application / delivery	Low	Production Function	127.52	127.52	N/A	Reported
Gibbons, D.C (26)	USA (Texas)	At site	Unknown	Application / delivery	Low	Production Function	277.12	277.12	N/A	Reported
Gibbons, D.C (26)	USA (Texas)	At site	Unknown	Application / delivery	Low	Production Function	85.83	85.83	N/A	Reported
Gibbons, D.C (26)	USA (Texas)	At site	Unknown	Application / delivery	Low	Production Function	139.78	139.78	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	High	Production Function	124.93	124.93	N/A	Reported
Gibbons, D.C (26)	USA (Arizona)	At site	Unknown	Application / delivery	High	Production Function	132.43	132.43	N/A	Reported
Gisser, M., Lansford, R.R., Gorman, W.D., Creel, B.J. and Evans B. (27)	USA (Four Corners Area)	At site	Unknown	Application / delivery	Unknown	Linear Programming	0 – 56.06	19.54 [i]	Median	Summarised
Hansen, L.T. and Hallam, A. (29)	USA (ASA 205)	Unknown	Unknown	Unknown	Unknown	Other	275.96	275.96	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 301)	Unknown	Unknown	Unknown	Unknown	Other	79.14	79.14	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 305)	Unknown	Unknown	Unknown	Unknown	Other	26.58	26.58	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 307)	Unknown	Unknown	Unknown	Unknown	Other	34.21	34.21	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 602)	Unknown	Unknown	Unknown	Unknown	Other	26.53	26.53	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 702)	Unknown	Unknown	Unknown	Unknown	Other	56.82	56.82	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 801)	Unknown	Unknown	Unknown	Unknown	Other	69.18	69.18	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Hansen, L.T. and Hallam, A. (29)	USA (ASA 803)	Unknown	Unknown	Unknown	Unknown	Other	17.19	17.19	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1007)	Unknown	Unknown	Unknown	Unknown	Other	5.42	5.42	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1008)	Unknown	Unknown	Unknown	Unknown	Other	59.22	59.22	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1010)	Unknown	Unknown	Unknown	Unknown	Other	21.16	21.16	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1101)	Unknown	Unknown	Unknown	Unknown	Other	11.23	11.23	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1103)	Unknown	Unknown	Unknown	Unknown	Other	66.48	66.48	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1104)	Unknown	Unknown	Unknown	Unknown	Other	152.71	152.71	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1201)	Unknown	Unknown	Unknown	Unknown	Other	29.62	29.62	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1203)	Unknown	Unknown	Unknown	Unknown	Other	42.43	42.43	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1304)	Unknown	Unknown	Unknown	Unknown	Other	33.28	33.28	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (ASA 1802)	Unknown	Unknown	Unknown	Unknown	Other	1,054.24	1,054.24	N/A	Reported
Hartman, L.M. and Anderson, R.L. (31)	USA (North-eastern Colorado)	In stream	Long	Application / delivery	Unknown	Hedonic	18.68	18.68	N/A	Reported
Houk, E. E., Frasier, M. and Taylor R.G. (34)	USA (Platte River Basin)	Unknown	Long	Application / delivery	Unknown	Other	9.88 – 88.82	36.25 [j]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	22.41 – 124.18	87.43 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	-117.93 – 37.30	6.72 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	0 – 49.04	49.04 [k]	Median	Summarised

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	15.37 – 47.58	20.65 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	0 -88.19	50.41 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	0 – 152.82	30.15 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	0 – 14.68	7.03 [k]	Median	Summarised
Howe, C.W. and Ahrens, W.A. (35)	USA (Upper Colorado River Basin)	At site	Long	Consumption	Low	Farm crop budget	46.36 – 77.40	69.44 [k]	Median	Summarised
Howe, C.W. and Easter, K.W. (36)	USA (Texas High Plains)	Unknown	Long	Unknown	Unknown	Other	122.73	122.73	N/A	Reported
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	7.36 – 46.59	26.98 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	7.36 – 68.67	38.01 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	66.21 – 85.83	76.02 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	73.57 – 78.48	76.02 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	61.31 – 100.55	80.93 [l]	Median	Summarised

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	Low	Linear Programming	166.76 – 213.35	190.06 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	High	Linear Programming	218.26 – 407.09	312.67 [l]	Median	Summarised
Kelso, M.M., Martin W.E. and Mack L.E. (39)	USA (Arizona - Roosevelt Water Conservancy District and the Salt River Project)	At site	Short	Application / delivery	High	Linear Programming	286.93	286.93	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Farm crop budget	36.79	36.79	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Farm crop budget	46.59	46.59	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Farm crop budget	139.78	139.78	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	Low	Farm crop budget	66.21	66.21	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	Low	Farm crop budget	78.48	78.48	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	Low	Farm crop budget	164.31	164.31	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Farm crop budget	63.76	63.76	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	Low	Farm crop budget	159.40	159.40	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	High	Farm crop budget	183.93	183.93	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	In stream	Long	Application / delivery	High	Farm crop budget	213.35	213.35	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	Low	Farm crop budget	98.09	98.09	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	Low	Farm crop budget	186.38	186.38	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	High	Farm crop budget	232.97	232.97	N/A	Reported
Lacewell, R.D., Sprott, J.M. and Beattie, B.R. (43)	USA (Texas – High Plains)	At site	Long	Application / delivery	High	Farm crop budget	247.69	247.69	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (California - Tudor Mutual and Feather Water Districts)	At site	Short	Withdrawal	Unknown	Water market transaction	12.22	12.22	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (Westlands Water District)	At site	Short	Withdrawal	Unknown	Water market transaction	73.34	73.34	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (Boise River Water Bank)	At site	Short	Withdrawal	Unknown	Water market transaction	8.96	8.96	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (SE Colorado)	In stream	Short	Application / delivery	Low	Farm crop budget	83.80	83.80	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (SE Colorado)	In stream	Short	Application / delivery	Low	Farm crop budget	40.45	40.45	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (SE Colorado)	In stream	Short	Application / delivery	Low	Farm crop budget	69.35	69.35	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Naeser, R.B. and Bennett, L.L. (52)	USA (SW Kansas)	In stream	Short	Application / delivery	Low	Farm crop budget	117.03	117.03	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (SW Kansas)	In stream	Short	Application / delivery	Low	Farm crop budget	78.02	78.02	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (SW Kansas)	In stream	Short	Application / delivery	Low	Farm crop budget	63.57	63.57	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-eastern Colorado)	At site	Long	Application / delivery	Low	Yield comparison	179.16	179.16	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-eastern Colorado)	At site	Long	Application / delivery	Low	Yield comparison	65.02	65.02	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-eastern Colorado)	At site	Long	Application / delivery	Low	Yield comparison	60.68	60.68	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-western Kansas)	At site	Long	Application / delivery	Low	Yield comparison	176.27	176.27	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-western Kansas)	At site	Long	Application / delivery	Low	Yield comparison	82.35	82.35	N/A	Reported
Naeser, R.B. and Bennett, L.L. (52)	USA (South-western Kansas)	At site	Long	Application / delivery	Low	Yield comparison	40.45	40.45	N/A	Reported
Renshaw, E.F. (57)	USA (California – San Diego)	At site	Unknown	Unknown	Unknown	Other	216.68	216.68	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Mississippi St Francis and Crittenden county)	At site	Short	Application / delivery	High	Farm crop budget	79.53	79.53	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Mississippi St Francis and Cittenden county)	At site	Short	Application / delivery	Low	Farm crop budget	98.98	98.98	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Mississippi St Francis and Crittenden county)	At site	Short	Application / delivery	High	Farm crop budget	82.31	82.31	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Quachita and Mississippi Tensas region)	At site	Short	Application / delivery	High	Farm crop budget	247.28	247.28	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Quachita and Mississippi Tensas region)	At site	Short	Application / delivery	Low	Farm crop budget	120.51	120.51	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas - Quachita and Mississippi Tensas region)	At site	Short	Application / delivery	High	Farm crop budget	104.89	104.89	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – White River region)	At site	Short	Application / delivery	High	Farm crop budget	203.52	203.52	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – White River region)	At site	Short	Application / delivery	Low	Farm crop budget	109.05	109.05	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – White River region)	At site	Short	Application / delivery	High	Farm crop budget	95.51	95.51	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – Lower Arkansas River and Benton county)	At site	Short	Application / delivery	High	Farm crop budget	203.52	203.52	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – Lower Arkansas River and Benton county)	At site	Short	Application / delivery	Low	Farm crop budget	109.05	109.05	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – Lower Arkansas River and Benton county)	At site	Short	Application / delivery	High	Farm crop budget	95.51	95.51	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – Lower Red River region)	At site	Short	Application / delivery	Low	Farm crop budget	117.39	117.39	N/A	Reported
Shulstad, R.N., Cross, E.D. and May, R.D. (59)	USA (Arkansas – Lower Red River region)	At site	Short	Application / delivery	High	Farm crop budget	101.07	101.07	N/A	Reported
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	Low	Linear Programming	53.95	53.95	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	Low	Linear Programming	63.76	63.76	N/A	Reported
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	Low	Linear Programming	63.76	63.76	N/A	Reported
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	High	Linear Programming	36.79 - 68.67	52.73 [m]	Median	Summarised
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	High	Linear Programming	61.31 – 100.55	80.93 [m]	Median	Summarised
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	High	Linear Programming	51.50 – 98.09	74.80 [m]	Median	Summarised
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	High	Linear Programming	90.74	90.74	N/A	Reported
Shumway, C.R. (60)	USA (California - West side of San Joaquin Valley)	At site	Long	Application / delivery	Low	Linear Programming	53.95	53.95	N/A	Reported
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (New Mexico)	In stream	Long	Application / delivery	Unknown	Hedonic	12.37 – 20.48	19.36 [n]	Median	Summarised
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (Oklahoma)	In stream	Long	Application / delivery	Unknown	Hedonic	2 – 6.79	2.85 [n]	Median	Summarised
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (Colorado – North)	In stream	Long	Application / delivery	Unknown	Hedonic	8.38 – 20.37	10.23 [n]	Median	Summarised
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (Colorado – South)	In stream	Long	Application / delivery	Unknown	Hedonic	3.16 – 15.88	4.75 [n]	Median	Summarised

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (Kansas)	In stream	Long	Application / delivery	Unknown	Hedonic	3.30 – 8.21	5.07 [n]	Median	Summarised
Torell, L.A., Libbin, J.D. and Miller, M.D. (61)	USA (Nebraska)	In stream	Long	Application / delivery	Unknown	Hedonic	2.77 – 7.84	6.70 [n]	Median	Summarised
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	High	Linear Programming	24.52	24.52	N/A	Reported
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	Low	Linear Programming	24.52	24.52	N/A	Reported
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	Low	Linear Programming	76.02	76.02	N/A	Reported
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	Low	Linear Programming	127.52	127.52	N/A	Reported
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	High	Linear Programming	191.28	191.28	N/A	Reported

Appendix 4. Agriculture (USA) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Washington State University Agricultural Research Centre (67)	USA (Washington)	At site	Unknown	Application / delivery	High	Linear Programming	210.90	210.90	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (North-eastern Colorado)	At site	Long	Withdrawal	Unknown	Water market transaction	58.82	58.82	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (South central Nebraska)	At site	Long	Application / delivery	Unknown	Yield comparison	53.77	53.77	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Eastern Washington)	At site	Long	Application / delivery	Unknown	Yield comparison	74.09	74.09	N/A	Reported

[a] Multiple marginal values reported for different experiment types and percentage reductions in water supply. [b] Multiple average values reported for percentage reductions in supply of water. [c] Median value in range given. [d] Multiple values reported for different discount rates and time spans. [e] Multiple values reported for different dryland yields and cotton prices. [f] Multiple values reported for the months between May and September and for a dry and normal year. [g] Multiple values reported for different land types. [h] Median value in range given. [i] Multiple values reported for various different elevations. [j] Multiple values reported for unidentified sub-regions. [k] Multiple values reported for individual crops in unidentified sub-regions. [l] Median value in range given. [m] Multiple values reported for individual crops in unidentified sub-regions. [n] Multiple values reported for each state between 1979 and 1986.

Appendix 5. Agriculture (USA) Water Right

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (Colorado – South Platte Basin)	At site	Long	Withdrawal	Unknown	Water market transaction	6,762.85	6,762.85 [a]	N/A	Reported
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (Nevada – Truckee Basin)	At site	Long	Withdrawal	Unknown	Water market transaction	2,684.74	2,684.74 [a]	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Arizona)	At site	Short	Withdrawal	Unknown	Water market transaction	1,311.44	1,311.44	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (California)	At site	Short	Withdrawal	Unknown	Water market transaction	1,570.77	1,570.77	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Colorado)	At site	Short	Withdrawal	Unknown	Water market transaction	2,703.30	2,703.30	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Idaho)	At site	Short	Withdrawal	Unknown	Water market transaction	108.60	108.60	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (New Mexico)	At site	Short	Withdrawal	Unknown	Water market transaction	2,089.88	2,089.88	N/A	Reported

Appendix 5. Agriculture (USA) Water Right

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Texas)	At site	Short	Withdrawal	Unknown	Water market transaction	394.88	394.88	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Utah)	At site	Short	Withdrawal	Unknown	Water market transaction	1,239.15	1,239.15	N/A	Reported
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Wyoming)	At site	Short	Withdrawal	Unknown	Water market transaction	221.44	221.44	N/A	Reported
Bustic, V. and Netrusil, N.R. (12)	USA (Oregon – Douglas County)	In stream	Long	Application/delivery	Unknown	Hedonic	339.14	339.14	N/A	Reported
Hartman, L.M. and Anderson, R.L. (32)	USA (Northeastern Colorado)	In stream	Long	Withdrawal	Unknown	Water market transaction	178.62	178.62	N/A	Reported

[a] Ditch company shares.

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Gross inflow	Low	Unclear	32.79	32.79	N/A	Reported
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Irrigation	Low	Unclear	49.18	49.18	N/A	Reported
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Consumption	Low	Unclear	65.57	65.57	N/A	Reported
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Gross inflow	Low	Unclear	98.36	98.36	N/A	Reported
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Irrigation	Low	Unclear	98.36	98.36	N/A	Reported
Ahmad, M., Masih, I. and Turral, H. (70)	Pakistan (Punjab)	Unknown	Unknown	Consumption	Low	Unclear	81.96	81.96	N/A	Reported
Al-Weshah, R. (72)	Jordan (jordan Valley)	Unknown	Unknown	Unknown	High	LP	905.50	905.50 [a]	N/A	Reported
Arias Rojo, R. H. (74)	Mexico (Saltillo, Coahuila)	Unknown	Unknown	Unknown	Unknown	Opportunity cost	392.06	392.06	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Consumption	Low	Farm crop budget	213.61	213.61 [b]	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Consumption	Low	Farm crop budget	207.84	207.84 [b]	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Consumption	Unknown	Farm crop budget	1,772.40	1,772.40 [b]	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Consumption	Unknown	Farm crop budget	2,141.89	2,141.89 [b]	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Withdrawal	Unknown	Farm crop budget	103.92	103.92	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Withdrawal	Unknown	Farm crop budget	184.75	184.75	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Total supply	Unknown	Farm crop budget	57.73	57.73	N/A	Reported
Bakker, M., Barker, R., Meinzen-Dick, R. and Knoradsen, F. (75)	Sri Lanka (Kirindi Oya Irrigation and Settlement Project)	Unknown	Unknown	Total supply	Unknown	Farm crop budget	92.37	92.37	N/A	Reported
Birol, E., Koundouri, P. and Kountouris, Y. (77)	Cyprus (Akrotiri aquifer)	Unknown	Unknown	Unknown	Unknown	Choice experiment	55.28	55.28 [c]	Mean	Reported
Birol, E., Koundouri, P. and Kountouris, Y. (78)	Cyprus (Limassol prefecture)	Unknown	Unknown	Unknown	Unknown	Contingent valuation	648.17 – 1,148.75	871.54 [d]	Median	Summarised
Bowen, R. and Young, R. (79)	Egypt (Kafr El Sheikh District)	Unknown	Long	Application	Unknown	LP	114.51	114.51 [e]	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Bowen, R. and Young, R. (79)	Egypt (Kafr El Sheikh District)	Unknown	Long	Application	Unknown	LP	146.24	146.24 [f]	Median	Summarised
Bowen, R. and Young, R. (79)	Egypt (Kafr El Sheikh District)	Unknown	Long	Application	Unknown	LP	37.94	37.94 [f]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	364.77	364.77 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	230.15	230.15 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	173.70	173.70 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Not classified (Canola)	Production function	191.07	191.07 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Not classified (Peas)	Production function	5.79	5.79 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	206.99	206.99 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	222.92	222.92 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	180.94	180.94 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	1648.71	1648.71 [g]	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	720.86	720.86 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	At site	Unknown	Application	N/A (weighted average Alberta)	Production function	277.92	277.92 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan)	At site	Unknown	Application	N/A (weighted average Saskatchewan)	Production function	292.40	292.40 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	N/A (weighted average Alberta and Saskatchewan)	Production function	279.37	279.37 [g]	N/A	Reported
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	330.03 - 398.07	363.33 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	196.86 – 214.23	205.55 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	192.51 – 264.89	228.71 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	167.91 – 179.49	173.70 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	428.46 – 628.22	528.34 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Not classified (Canola)	Production function	163.57 – 217.12	189.62 [h]	Median	Summarised

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	154.88 – 206.99	180.94 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	179.49 – 259.1	219.3 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	1.45 – 10.13	5.79 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	1,252.1 – 2,043.88	1,648.71 [h]	Median	Summarised
Bruneau, J. (80)	Canada (Saskatchewan and Alberta)	At site	Unknown	Application	Low	Production function	502.29 – 939.43	720.86 [h]	Median	Summarised
Calatrava, L.J. and Sayadi, S. (81)	Spain (Granada Coast)	Unknown	Unknown	Consumption	High	CV	603.07	603.07	Mean	Reported
El Chami, C.E., Knox, J.W., Daccache, A. and Weatherhead, E.K. (82)	UK (East of England)	At site	Different scenarios	Consumption	Low	Yield comparison	140.94	140.94 [i]	N/A	Reported
El Chami, C.E., Knox, J.W., Daccache, A. and Weatherhead, E.K. (82)	UK (East of England)	At site	Different scenarios	Consumption	Low	Yield comparison	422.81	422.81 [i]	N/A	Reported
El Chami, C.E., Knox, J.W., Daccache, A. and Weatherhead, E.K. (82)	UK (East of England)	At site	Different scenarios	Consumption	Low	Yield comparison	563.75	563.75 [i]	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Emerton, L., Erdenesaikhan, N., De Veen, B., Tsogoo, D., Janchivdorj, L., Suvd, P., Enkhtsetseg, B., Gandolgor, G., Dorisuren, C., Sainbayar, D. and Enkhbaatar, A. (84)	Mongolia	Unknown	Unknown	Unknown	High	WTP inflator	17,441.36	17,441.36 [j]	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	Not classified (Grape)	Farm crop budget	45.128	45.128	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	High	Farm crop budget	1,051.25	1,051.25	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	Not classified (Pomegranate)	Farm crop budget	15.84	15.84	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	Not classified (Almond)	Farm crop budget	368.28	368.28	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	High	Farm crop budget	3,271.40	3,271.40	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	High	Farm crop budget	285.68	285.68	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	Low	Farm crop budget	626.96	626.96	N/A	Reported
Esmaeili, A. and Vazirzadeh, S. (85)	Iran (Essin in Hormozgan province)	Unknown	Unknown	Unknown	Not classified (Onion)	Farm crop budget	1,305.50	1,305.50	N/A	Reported
Hellegers, P.J.G.J. and Perry, C.J. (86)	Egypt (Kemry)	At site	Short	Application	Low	Farm crop budget	120.50	120.50	N/A	Reported
Hellegers, P.J.G.J. and Perry, C.J. (86)	Morocco (Tadla)	At site	Short	Application	Low	Farm crop budget	150.63	150.63	N/A	Reported
Hellegers, P.J.G.J. and Perry, C.J. (86)	India (Haryana)	At site	Short	Application	Low	Farm crop budget	60.25	60.25	N/A	Reported
Hellegers, P.J.G.J. and Perry, C.J. (86)	Indonesia (Brantas)	At site	Short	Application	Low	Farm crop budget	60.25	60.25	N/A	Reported
Hellegers, P.J.G.J. and Perry, C.J. (86)	Ukraine (Crimea)	At site	Short	Application	Low	Farm crop budget	165.69	165.69	N/A	Reported
Hussain, I., Turral, H., Molden, D. and Ahmad, M.D. (89)	Pakistan (Indus Basin)	Unknown	Unknown	Application	Not classified (Variety)	Unclear	63.14	63.14	N/A	Reported
Kadigi, R., Mdoe, N., Oshimogo, G. and Moradet, S. (90)	Tanzania (Great Ruaha River Catchment)	At source	Short	Withdrawn	Low	Yield comparison	15.48	15.48	N/A	Reported
Kadigi, R., Mdoe, N., Oshimogo, G. and Moradet, S. (90)	Tanzania (Great Ruaha River Catchment)	At source	Short	Consumed	Low	Yield comparison	61.91	61.91	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Low	Farm crop budget	488.86	488.86	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Low	Farm crop budget	141.36	141.36	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Not classified (Cowpeas)	Farm crop budget	9.86	9.86	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Not classified (Green grams)	Farm crop budget	685.45	685.45	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Not classified (Cassava)	Farm crop budget	41.09	41.09	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Not classified (Banana)	Farm crop budget	44.71	44.71	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Not classified (Mangoes)	Farm crop budget	29.59	29.59	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	High	Farm crop budget	14.79	14.79	N/A	Reported
Kiprop, J.K., Lagat, J.K., Mshenga, P. and Macharia, A.M. (92)	Kenya (Kerio Valley Basin)	At site	Short	Application	Low	Farm crop budget	370.84	370.84	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	Low	Yield comparison	4,537.53	4,537.53	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	Low	Yield comparison	7,450.84	7,450.84	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	Low	Yield comparison	825.01	825.01	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	High	Yield comparison	1,856.26	1,856.26	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	High	Yield comparison	3,222.68	3,222.68	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	High	Yield comparison	4,614.88	4,614.88	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	Low	Yield comparison	77.34	77.34	N/A	Reported
Knox, J.W., Morris, E.K., Weatherhead, E.K. and Turner, A.P. (93)	UK (Anglian Region)	Unknown	Unknown	Application	Low	Yield comparison	77.34	77.34	N/A	Reported
Kulshreshtha, S.N. (94)	Canada (Manitoba)	Unknown	Short	Unknown	Low	Unclear	995.94	995.94	N/A	Reported
Kulshreshtha, S.N. (94)	Canada (Manitoba)	Unknown	Short	Unknown	Unknown	Unclear	321.08	321.08	N/A	Reported
Kulshreshtha, S.N. and W.J. Brown (95)	Canada (Saskatchewan)	Unknown	Short	Unknown	Low	Yield comparison	113.04	113.04 [k]	Mean	Reported
Kulshreshtha, S.N. and W.J. Brown (95)	Canada (Saskatchewan)	Unknown	Long	Unknown	Low	Yield comparison	38.33	38.33 [k]	Mean	Reported
Latinopoulos, P., Tziakas, V. and Mallios, Z. (98)	Greece (Chalkidiki)	At source	Long	Withdrawal	Unknown	HPM	143.45	143.45 [l]	N/A	Reported
Louw, D.B. and van Schalkwyk, H.D. (100)	South Africa (Oilfants River Basin)	Unknown	Unknown	Withdrawal	High	LP	337.22	337.22 [m]	N/A	Reported
Martinez-Paz, J. M. and Perni, A. (101)	Spain (Segura Basin District)	Unknown	Unknown	Consumption	High	Yield comparison	703.23	703.23	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Menegaki, A. N., Hanley, N. and Tsagarakis, K.P. (103)	Greece (Crete)	Unknown	Unknown	Unknown	Not classified (Olive trees and tomato crops)	CV	287.84	287.84 [n]	Mean	Reported
Moran, D and Dann, S. (106)	England and Scotland	At site	Unknown	Unknown	Low	Net back analysis	527.59 – 3,165.52	1,846.55 [o]	Median	Summarised
Muller, R.A. (107)	Canada	At site	Unknown	Unknown	Unknown	Benefit transfer	0.00	0.00 [p]	N/A	Reported
Muller, R.A. (107)	Canada	At site	Unknown	Unknown	Unknown	Benefit transfer	70.18	70.18 [p]	N/A	Reported
Nieuwoudt, W.L., Backeberg, G.R. and Du Pleiss, H.M. (108)	South Africa	Unknown	Unknown	Unknown	Unknown	Benefit Transfer	0 – 60.96	30.48 [q]	Median	Summarised
Pazvakawambwa, G.T., Van Der Zaag, P. (119)	Zimbabwe (Nyanyadzi smallholder irrigation scheme)	Unknown	Unknown	Unknown	Low	Unclear	245.89	245.89 [r]	N/A	Reported
Pazvakawambwa, G.T., Van Der Zaag, P. (109)	Zimbabwe (Nyanyadzi smallholder irrigation scheme)	Unknown	Unknown	Unknown	Low	Unclear	311.47	311.47 [r]	N/A	Reported
Puente Gonzalez, A. (111)	Mexico (Cuenca Alta del Río La Antigua-Veracruz)	Unknown	Unknown	Unknown	Low	Unclear	183.94	183.94	N/A	Reported
Puente Gonzalez, A. (111)	Mexico (Cuenca Alta del Río La Antigua-Veracruz)	Unknown	Unknown	Unknown	Low	Unclear	127.63	127.63	N/A	Reported

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Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Qureshi, M. E., Connor, J., Kirby, M. and Mainuddin, M. (112)	Australia (Murray Basin)	Unknown	Short	Application	Different in different sub-basins	Mathematical Programming	7.25 – 188.58	16.92 [s]	Median	Summarised
Renwick, M.E. (114)	Sri Lanka (Kirindi Oya irrigation system)	At site	Unknown	Delivered	Low	Farm crop budget	107.51	107.51 [t]	Mean	Reported
Rodgers, C. and Hellegers, P.J.G.J. (116)	Indonesia (Brantas)	At site	Short	Application	Low	Farm crop budget	32.79 – 81.97	57.375 [u]	Median	Summarised
Rodgers, C. and Hellegers, P.J.G.J. (116)	Indonesia (Brantas)	At site	Short	Application	Low	Farm crop budget	131.14 – 180.32	155.73 [u]	Median	Summarised
Rodgers, C. and Hellegers, P.J.G.J. (116)	Indonesia (Brantas)	At site	Short	Application	High	Farm crop budget	65.57 – 81.97	73.77 [u]	Median	Summarised
Rodgers, C. and Hellegers, P.J.G.J. (116)	Indonesia (Brantas)	At site	Short	Application	Not classified (Groundnuts)	Farm crop budget	65.57 – 131.14	98.36 [u]	Median	Summarised
Rogers, P., Bhatia, R. and Huber, A. (117)	India (Haryana)	At site	Short	Withdrawn	Low	Farm crop budget	36.14	36.14	N/A	Reported
Rogers, P., Bhatia, R. and Huber, A. (117)	India (Jamshedpur, Subernarekha River Basin)	At site	Short	Withdrawn	Low	Farm crop budget	51.36	51.36	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Short	Application	Unknown	Yield comparison	65.17	65.17 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Short	Application	Unknown	Yield comparison	48.1	48.1 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Short	Application	Unknown	Yield comparison	52.48	52.48 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Short	Application	Unknown	Yield comparison	56.38	56.38 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Long	Application	Unknown	Yield comparison	30.43	30.43 [v]	N/A	Reported

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Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Long	Application	Unknown	Yield comparison	21.97	21.97 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Long	Application	Unknown	Yield comparison	22.06	22.06 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Alberta)	Unknown	Long	Application	Unknown	Yield comparison	20.71	20.71 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Saskatchewan)	Unknown	Short	Application	Unknown	Yield comparison	99.51	99.51 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Saskatchewan)	Unknown	Short	Application	Unknown	Yield comparison	19.62	19.62 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Saskatchewan)	Unknown	Long	Application	Unknown	Yield comparison	75.44	75.44 [v]	N/A	Reported
Samarawickrema, A. and Kulshreshtha, S. (118)	Canada (Saskatchewan)	Unknown	Long	Application	Unknown	Yield comparison	11.71	11.71 [v]	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Low	Yield comparison	129.53	129.53	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Low	Yield comparison	154.59	154.59	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Not classified (Butternuts)	Yield comparison	300.83	300.83	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Low	Farm crop budget	146.24	146.24	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Not classified (Cabbage intensive)	Farm crop budget	685.23	685.23	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Low	Farm crop budget	16.71	16.71	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	**Country (location) (2)**	**In-stream/at source (3)**	**Short/long run (4)**	**Volumetric measure (5)**	**Crop value (6)**	**Valuation approach (7)**	**Value range 2014 $/AF (8)**	**2014 $/AF (9)**	**Measure of central tendency (10)**	**Reported/ summarised (11)**
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Not classified (Cabbage extensive)	Farm crop budget	37.60	37.60	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Not classified (Butternut high yield)	Farm crop budget	313.37	313.37	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	In stream	Short	Unknown	Not classified (Butternet low yield)	Farm crop budget	8.36	8.36	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Low	Farm crop budget	83.56	83.56	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Low	Farm crop budget	902.50	902.50	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Not classified (Cabbage intensive)	Farm crop budget	476.32	476.32	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Low	Farm crop budget	0.000	0.000	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Low	Farm crop budget	208.91	208.91	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	In stream	Short	Unknown	Not classified (Cabbage extensive)	Farm crop budget	380.22	380.22	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	Unknown	Unknown	Unknown	Unknown	CV	12.54	12.54 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	Unknown	Unknown	Unknown	Unknown	CV	8.36	8.36 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	Unknown	Unknown	Unknown	Unknown	CV	4.18	4.18 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Zanyokwe - Eastern Cape)	Unknown	Unknown	Unknown	Unknown	CV	0.000	0.000 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	Unknown	Unknown	Unknown	Unknown	CV	8.36	8.36 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	Unknown	Unknown	Unknown	Unknown	CV	12.54	12.54 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	Unknown	Unknown	Unknown	Unknown	CV	79.39	79.39 [w]	N/A	Reported
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	Unknown	Unknown	Unknown	Unknown	CV	12.54	12.54 [w]	N/A	Reported

Appendix 6. Agriculture (Rest of the World) Per Period

Author (paper reference number) (1)	Country (location) (2)	In-stream/at source (3)	Short/long run (4)	Volumetric measure (5)	Crop value (6)	Valuation approach (7)	Value range 2014 $/AF (8)	2014 $/AF (9)	Measure of central tendency (10)	Reported/ summarised (11)
Yokwe, S. (125)	South Africa (Thabina - Limpopo Province)	Unknown	Unknown	Unknown	Unknown	CV	0.000	0.000 [w]	N/A	Reported
Zetina-Espinosa, A. M., Mora-Flores, J. S., Martínez-Damián,M. A., Cruz-Jiménez, J. and Téllez-Delgado, R. (126)	Mexico (State of Mexico, Hidalgo)	At site	Unknown	Unknown	Unknown	Linear programming	173.44 – 1,033.44	603.44 [x]	Median	Summarised
Zetina-Espinosa, A. M., Mora-Flores, J. S., Martínez-Damián,M. A., Cruz-Jiménez, J. and Téllez-Delgado, R. (126)	Mexico (State of Mexico, Hidalgo)	At site	Unknown	Unknown	Unknown	Linear programming	5.42 – 37.94	21.68 [y]	Median	Summarised

[a] Value for existing use not proposed scenarios. [b] Volumetric measure refers to evapotranspiration (consumption) including rain water. [c] WTP to maintain water quantity by replenishing a threatened aquifer with treated wastewater. [d] Recycled wastewater use. Median value across different farm profiles and proposed wastewater use programmes. [e] Estimated present average financial return. Note: values were updated using 1990 PPP conversion rates, rather than 1986 rates as required, as the World Bank does not provide PPP data prior to 1990. [f] Median value across four water reduction scenarios (10,20,30 and 40%). Note: values were updated using 1990 PPP conversion rates, rather than 1986 rates as required, as the World Bank does not provide PPP data prior to 1990. [g] Average value. [h] Median value across different levels of water application. [i] Different scenarios modelled: 1) existing irrigation system no reservoir 2) existing irrigation system and reservoir, and 3) investment in new irrigation system but no reservoir. [j] Note: source makes reference to nominal exchange rate of 1,500 TUG to 1 USD. However, the world bank PPP rate in 2009 was 346 TUG to 1 USD which explains, in part, the very large value recorded here and demonstrates the sensitivity of the value estimates to the precise conversion rate used. However, even using the nominal rate quoted in the source, the value per acre foot would still be $4,027 (2014 USD). [k] Note: values were updated using 1990 PPP conversion rates, rather than 1986 rates as required, as the World Bank does not provide PPP data prior to 1990. [l] Marginal vale for one-time extraction. [m] Base analysis scenario reflecting current situation. [n] Recycled wastewater use. [o] Median value within range given. [p] Note: values were updated using 1990 PPP conversion rates, rather than 1986 rates as required, as the World Bank does not provide PPP data prior to 1990. [q] Median value within range given. [r] Lower value represents value of irrigation and rainfall used in crop production; upper value represents value of irrigation only. [s] Median value across different sub-basins of the Murray basin (baseline scenario). [t] Average value across two areas of interest in the study. [u] Median value within range given. [v] Each value estimate is for a different sub-basin in Alberta or Saskatchewan. [w] Different WTP values are given for different farm types and land users. [x] Autumn/winter cycle. Median value within the range cited in EVRI. [y] Spring/summer cycle. Median value within the range cited in EVRI.

Appendix 7. Industry (USA)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Food products	Added value	22,745.74	22,745.74 [a]	N/A	Reported
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Chemicals and stone	Added value	4,680.98	4,680.98 [a]	N/A	Reported
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Clay and glass	Added value	15,920.11	15,920.11 [a]	N/A	Reported
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Food products	Added value	1,819.85	1,819.85 [b]	N/A	Reported
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Chemicals and stone	Added value	420.33	420.33 [b]	N/A	Reported
D'Arge, R.C. (19)	USA (New Mexico and Texas)	Clay and glass	Added value	797.68	797.68 [b]	N/A	Reported
Kane, J. and Osantowski, R. (38)	USA (Geographically non-specific)	Meat packing	Alternative cost (chemical clarification, filtration, carbon absorption and reverse osmosis desalination)	733.45 – 1,022.79	878.12 [c]	Median	Summarised
Kneese, A.V. (40)	USA (Geographically non-specific)	Petroleum and coal products	Added value	41,568.92	41,568.92 [d]	N/A	Reported
Kneese, A.V. (40)	USA (Geographically non-specific)	Printing and publishing	Added value	1,736,020.32	1,736,020.32 [d]	N/A	Reported
Kneese, A.V. (41)	USA (New Mexico)	Manufacturing	Residual imputation	924.80 – 2,134.16	1,529.48 [e]	Median	Summarised
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Industrial water cooling	Cost of intake	48.22	48.22 [f]	N/A	Reported

Appendix 7. Industry (USA)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Industrial water cooling	Cost of intake	75.03	75.03 [g]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (cotton textile mill)	Cost of intake	364.45	364.45 [f]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (cotton textile mill)	Cost of intake	1,049.36	1,049.36 [g]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (unbleached Kraft paper mill)	Cost of intake	91.12	91.12 [f]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (unbleached Kraft paper mill)	Cost of intake	169.35	169.35 [g]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (basic oxygen steelmaking operations)	Cost of intake	125.40	125.40 [f]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (basic oxygen steelmaking operations)	Cost of intake	434.12	434.12 [g]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (cotton textile finishing)	Alternative cost (carbon absorption treatment)	300.14	300.14 [h]	N/A	Reported
Kollar, K.L., Brewer, R. and McAuley, P.H. (42)	USA (Geographically non-specific)	Process water (cotton textile finishing)	Alternative cost (carbon absorption treatment and demineralisation)	1,414.90	1,414.90 [i]	N/A	Reported
Lofting, E.M. and McGaughey, P.H. (45)	USA (California)	Cotton sector	Added value	457.26	457.26 [j]	N/A	Reported
Lofting, E.M. and McGaughey, P.H. (45)	USA (California)	Cotton sector	Added value	553.53	553.53 [k]	N/A	Reported
Lofting, E.M. and McGaughey, P.H. (45)	USA (California)	Textile products	Added value	2,214,103.49	2,214,103.49 [j]	N/A	Reported
Lofting, E.M. and McGaughey, P.H. (45)	USA (California)	Crude petroleum and natural gas	Added value	117,413.67	117,413.67 [k]	N/A	Reported

Appendix 7. Industry (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Industry sector / purpose (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported/ summarised (8)**
Powell, S.T. (56)	USA (Geographically non-specific)	National average for all industry	Added value	27,373.65	27,373.65	N/A	Reported
Renshaw, E.F. (57)	USA (Geographically non-specific)	Industry (Nonspecific)	Cost of intake	260.72	260.72	Mean	Reported
Wollman, N. et al. (68)	USA (New Mexico)	Mining and manufacturing sector	Added value	7,909.90 – 24,338.14	16,124.02 [l]	Median	Summarised
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (six unspecified regions)	Industrial water cooling	Alternative cost (water recirculation)	10.85 – 18.26	14.14 [m]	Median	Summarised
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Industrial water cooling (thermal power generation with low cost coal)	Alternative cost (water recirculation)	10.36	10.36	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Industrial water cooling (thermal power generation with medium cost coal)	Alternative cost (water recirculation)	11.02	11.02	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Industrial water cooling (thermal power generation with high cost coal)	Alternative cost (water recirculation)	11.65	11.65	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (East region)	Industrial water cooling (thermal power generation)	Alternative cost (water recirculation)	16.04	16.04	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (West region)	Industrial water cooling (thermal power generation)	Alternative cost (water recirculation)	11.50	11.50	N/A	Reported

Appendix 7. Industry (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Industry sector / purpose (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported/ summarised (8)**
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Arkansas, Louisiana, Oklahoma, Texas)	Industrial water cooling (thermal power generation)	Alternative cost (water recirculation)	15.19	15.19	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (West south central region)	Industrial water cooling (thermal power generation)	Alternative cost (water recirculation)	8.80	8.80	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Industrial water cooling (petroleum industry)	Alternative cost (water recirculation)	24.28	24.28	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Industrial water cooling (sugar beet industry)	Alternative cost (water recirculation)	34.07 – 39.04	36.55 [n]	Median	Summarised
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Maryland)	Process water (steel industry)	Alternative cost (water recirculation)	56.77	56.77	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (California)	Process water (steel industry)	Alternative cost (water recirculation)	21.31	21.31	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Arizona)	Process water (mineral industry)	Alternative cost (water recirculation)	14.20 – 28.41	21.31 [o]	Median	Summarised
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Geographically non-specific)	Process water (paper industry)	Alternative cost (water recirculation)	113.54	113.54	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	USA (Great plains)	Process water (paper industry – flume water)	Alternative cost (water recirculation)	161.86	161.86	N/A	Reported

[a] As reported in Young and Gray (1972), these estimates refer to value added. [b] As reported in Young and Gray (1972), these estimates refer to incremental value added. However, it is not clear how incremental value added has been calculated, and therefore, how it is distinct from the value-added measure. [c] Median value in range given. [d] As reported in Young and Gray (1972), it appears that the estimates from Kneese include secondary multiplier effects. [e] Median value in range given. [f] Total cost of gross water applied (no control option). Assumes that plant minimises costs and does not consider environmental impact. [g] Total cost of gross water applied (best available treatment option). [h] Price at which it becomes cost effective to introduce carbon absorption treatment for dyes and thereby increase water recycling to 76% of gross demand. [i] Price at which it would also become cost effective to introduce demineralisation and recycle the remaining 9% of non-

consumptive water withdrawals. [j] Direct value added only. [k] Direct and indirect value added. [l] Median value in range given. [m] Median value across six unspecified regions. [n] Median value in range given. [o] Median value in range given.

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Bruneau, J. (80)	Canada (Alberta)	Resource industries	Added value	15,994.97	15,994.97	Unclear	Reported
Bruneau, J. (80)	Canada (Alberta)	Manufacturing	Added value	30,542.43	30,542.43	Unclear	Reported
Bruneau, J. (80)	Canada (Alberta)	Services	Added value	21,003.35	21,003.35	Unclear	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Food)	Alternative cost	438.60	438.60	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Beverages)	Alternative cost	600.72	600.72	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Rubber products)	Alternative cost	623.88	623.88	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Plastic products)	Alternative cost	1,370.79	1,370.79	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Primary textiles)	Alternative cost	112.91	112.91	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Textile products)	Alternative cost	2,663.42	2,663.42	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Wood products)	Alternative cost	442.94	442.94	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Paper and allied products)	Alternative cost	86.85	86.85	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Primary metals)	Alternative cost	137.51	137.51	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Fabricated metals)	Alternative cost	671.64	671.64	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Transportation equipment)	Alternative cost	877.19	877.19	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Non-metallic mineral products)	Alternative cost	212.78	212.78	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Process water (Petroleum and coal products)	Alternative cost	108.56	108.56	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Industry sector / purpose (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported/ summarised (8)**
Bruneau, J. (80)	Canada (Alberta)	Process water (Chemicals and chemical products)	Alternative cost	94.09	94.09	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Food)	Alternative cost	393.72	393.72	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Beverages)	Alternative cost	502.29	502.29	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Rubber products)	Alternative cost	541.37	541.37	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Plastic products)	Alternative cost	1,230.38	1,230.38	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Primary textiles)	Alternative cost	89.75	89.75	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Textile products)	Alternative cost	2,595.38	2,595.38	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Wood products)	Alternative cost	398.06	398.06	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Paper and allied products)	Alternative cost	59.35	59.35	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Primary metals)	Alternative cost	108.56	108.56	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Fabricated metals)	Alternative cost	612.30	612.30	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Transportation equipment)	Alternative cost	744.02	744.02	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Non-metallic mineral products)	Alternative cost	193.97	193.97	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Petroleum and coal products)	Alternative cost	53.56	53.56	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Bruneau, J. (80)	Canada (Alberta)	Raw intake water (Chemicals and chemical products)	Alternative cost	63.69	63.69	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Food)	Alternative cost	3,597.06	3,597.06	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Beverages)	Alternative cost	2,175.61	2,175.61	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Rubber products)	Alternative cost	6,658.54	6,658.54	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Plastic products)	Alternative cost	12,581.75	12,581.75	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Primary textiles)	Alternative cost	3,721.54	3,721.54	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Textile products)	Alternative cost	18,542.59	18,542.59	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Wood products)	Alternative cost	1,483.70	1,483.70	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Paper and allied products)	Alternative cost	664.41	664.41	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Primary metals)	Alternative cost	1,283.94	1,283.94	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Fabricated metals)	Alternative cost	10,791.18	10,791.18	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Transportation equipment)	Alternative cost	2,559.20	2,559.20	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Non-metallic mineral products)	Alternative cost	1,033.52	1,033.52	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Petroleum and coal products)	Alternative cost	887.32	887.32	N/A	Reported
Bruneau, J. (80)	Canada (Alberta)	Consumed water (Chemicals and chemical products)	Alternative cost	788.89	788.89	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Industry sector / purpose (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported/ summarised (8)**
Emerton, L., Erdenesaikhan, N., De Veen, B., Tsogoo, D., Janchivdorj, L., Suvd, P., Enkhtsetseg, B., Gandolgor, G., Dorisuren, C., Sainbayar, D. and Enkhbaatar, A. (84)	Mongolia	Industrial and commercial	Benefit transfer	1,914.67	1,914.67 [a]	N/A	Reported
Kulshreshtha, S.N. (94)	Canada (Manitoba)	Manufacturing	Opportunity cost	1,056.17	1,056.17	Mean	Reported
Kumar, S. (96)	India (Geographically non-specific)	Leather industry	Input distance function	190.73	190.73	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Distillery	Input distance function	1,109.24	1,109.24	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Chemicals	Input distance function	519.79	519.79	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Sugar	Input distance function	798.75	798.75	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Paper and paper products	Input distance function	5,016.40	5,016.40	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Fertilizer	Input distance function	404.96	404.96	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Drug and pharmaceuticals	Input distance function	643.83	643.83	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Petrochemicals	Input distance function	229.34	229.34	N/A	Reported
Kumar, S. (96)	India (Geographically non-specific)	Miscellaneous	Input distance function	497.12	497.12	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Industry sector / purpose (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported/ summarised (8)**
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Refined petrol/coal)	Cost function	452.44	452.44	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Primary metal)	Cost function	168.09	168.09	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Chemicals)	Cost function	113.11	113.11	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Fabricated metals)	Cost function	75.41	75.41 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Beverages)	Cost function	59.70	59.70 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Plastic)	Cost function	50.27	50.27 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Paper)	Cost function	48.70	48.70 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Transport equipment)	Cost function	39.27	39.27 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Non-metallic minerals)	Cost function	36.13	36.13 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Wood)	Cost function	31.42	31.42 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Textile products)	Cost function	7.85	7.85 [b]	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Primary textiles)	Cost function	2.2	2.2 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Rubber)	Cost function	9.43	9.43 [b]	N/A	Reported
Renzetti, S. and Dupont, D.P. (115)	Canada (Geographically non-specific)	Intake water (Food)	Cost function	26.71	26.71 [b]	N/A	Reported
Rogers, P., Bhatia, R. and Huber, A. (117)	India (Jamshedpur, Subernarekha River Basin)	Industry unspecified	Added value	4,425.83	4,425.83	N/A	Reported
Tan, R. P. and Bautista, G.M. (119)	Philippines (Cagayan de Oro)	Industry unspecified	Unclear	16.39	16.39	Unclear	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Coal mining	Production function	1,038.96	1,038.96	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Petroleum extraction	Production function	5,436.64	5,436.64	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Metal mining and preparation	Production function	806.09	806.09	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Food and beverage manufacturing	Production function	2,301.84	2,301.84	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Textiles	Production function	10,300.06	10,300.06	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Paper and pulp products	Production function	752.35	752.35	N/A	Reported

Appendix 8. Industry (Rest of the World)

Author (paper reference number) (1)	Country (location) (2)	Industry sector / purpose (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Power generation	Production function	44.78	44.78	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Petroleum	Production function	4,863.42	4,863.42	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Chemicals	Production function	877.74	877.74	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Medical products	Production function	2,919.84	2,919.84	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Construction	Production function	4,926.12	4,926.12	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Smelting	Production function	3,421.41	3,421.41	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Industrial equipment and machinery	Production function	7,971.35	7,971.35	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Transportation equipment	Production function	24,030.50	24,030.50	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Electronic equipment	Production function	21,863.00	21,863.00	N/A	Reported
Wang, H and Lall, S. (122)	China (Geographically non-specific)	Leather goods	Production function	15,638.18	15,638.18	N/A	Reported
Young, R.A. and Gray, S.L. with Held, R.b. and Mack, R.S. (69)	Mexico (Monterey)	Process water (chemical industry)	Alternative cost (water recirculation)	99.38	99.38	N/A	Reported

[a] Value calculated by inflating local tariffs by estimates of WTP in a World Bank study. [b] Coefficients for these uses were not statistically significant.

Appendix 9. Municipal (USA)

Author (paper reference number) (1)	Country (location) (2)	Domestic specific? (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported /summarised (8)	Elasticity (where available) (9) *
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (California – Central valley)	NO (Municipal and industrial)	Water market transaction	214.78	214.78 [a]	N/A	Reported	
Booker, J.F. and Colby, B.G. (6)	USA (Colorado – Denver)	NO (Municipal)	Demand function	701.83	701.83 [b]	N/A	Reported	-0.45
Booker, J.F. and Colby, B.G. (6)	USA (Utah – Central Utah Project)	NO (Municipal)	Demand function	699.98	699.98 [b]	N/A	Reported	-0.45
Booker, J.F. and Colby, B.G. (6)	USA (New Mexico – Albuquerque)	NO (Municipal)	Demand function	739.92	739.92 [b]	N/A	Reported	-0.38
Booker, J.F. and Colby, B.G. (6)	USA (Nevada – Las Vegas)	NO (Municipal)	Demand function	622.87	622.87 [b]	N/A	Reported	-0.44
Booker, J.F. and Colby, B.G. (6)	USA (Arizona – Central Arizona)	NO (Municipal)	Demand function	559.65	559.65 [b]	N/A	Reported	-0.43
Booker, J.F. and Colby, B.G. (6)	USA (California – Southern California)	NO (Municipal)	Demand function	530.35	530.35 [b]	N/A	Reported	-0.38
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Arizona)	NO (Urban-municipal and industrial)	Water market transaction	99.61	99.61 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (California)	NO (Urban-municipal and industrial)	Water market transaction	151.58	151.58 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Colorado)	NO (Urban-municipal and industrial)	Water market transaction	52.16	52.16 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Idaho)	NO (Urban-municipal and industrial)	Water market transaction	4.36	4.36 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Montana)	NO (Urban-municipal and industrial)	Water market transaction	31.98	31.98 [c]	N/A	Reported	

Appendix 9. Municipal (USA)

Author (paper reference number) (1)	Country (location) (2)	Domestic specific? (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported /summarised (8)	Elasticity (where available) (9) *
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Nevada)	NO (Urban-municipal and industrial)	Water market transaction	43.20	43.20 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Oregon)	NO (Urban-municipal and industrial)	Water market transaction	11.79	11.79 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Texas)	NO (Urban-municipal and industrial)	Water market transaction	34.20	34.20 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Utah)	NO (Urban-municipal and industrial)	Water market transaction	167.46	167.46 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Washington)	NO (Urban-municipal and industrial)	Water market transaction	45.45	45.45 [c]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Wyoming)	NO (Urban-municipal and industrial)	Water market transaction	91.96	91.96 [c]	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	YES	Water market transaction	21.73	21.73	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	NO (Municipal)	Water market transaction	21.24	21.24	N/A	Reported	
Gibbons, D.C. (26)	USA (Arizona -Tuscon)	YES	Demand function	9.81 – 551.78	77.25 [d]	Median	Summarised	-0.23 [f] -0.70 [g]
Gibbons, D.C. (26)	USA (North Carolina – Raleigh)	YES	Demand function	0 – 877.94	66.21 [d]	Median	Summarised	-0.305 [f] -1.380 [g]
Young, R.A. and Gray, S.L. with Held, R.B. and Mack, R.S. (69)	USA (East)	YES (Lawn sprinkler)	Demand function	71.02	71.02 [e]	N/A	Reported	-1.6
Young, R.A. and Gray, S.L. with Held, R.B. and Mack, R.S. (69)	USA (West)	YES (Lawn sprinkler)	Demand function	270.13	270.13 [e]	N/A	Reported	-0.7

Appendix 9. Municipal (USA)

Author (paper reference number) (1)	Country (location) (2)	Domestic specific? (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported /summarised (8)	Elasticity (where available) (9) *
Young, R.A. and Gray, S.L. with Held, R.B. and Mack, R.S. (69)	USA (East and West)	YES (In-house consumption)	Demand function	440.05	440.05 [e]	N/A	Reported	-0.25

* Price elasticities noted in each paper were not necessarily used to derive the corresponding value estimates in all cases. [a] Short-term lease value. [b] It is unclear whether this value refers to total value or net consumer surplus. Value refers to full use. [c] Agriculture to urban lease price. [d] Values refer to net consumer surplus i.e. the value of the water at source net of water utility costs. Values are derived for various reductions in the average monthly consumption. [e] Values refer to net consumer surplus i.e. the value of the water at source net of water utility costs. [f] Winter price elasticity. [g] Summer price elasticity.

Appendix 10. Municipal (USA) Water Right

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (Colorado – South Platte Basin)	NO (Urban)	Water market transaction	9,467.99	9,467.99 [a]	N/A	Reported	
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (Nevada – Truckee Basin)	NO (Urban)	Water market transaction	21,477.95	21,477.95	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Arizona)	NO (Urban -municipal and industrial)	Water market transaction	331.78	331.78 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (California)	NO (Urban -municipal and industrial)	Water market transaction	1,165.70	1,165.70 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Colorado)	NO (Urban -municipal and industrial)	Water market transaction	6,700.54	6,700.54 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Idaho)	NO (Urban -municipal and industrial)	Water market transaction	193.30	193.30 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (New Mexico)	NO (Urban -municipal and industrial)	Water market transaction	2,894.68	2,894.68 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Nevada)	NO (Urban -municipal and industrial)	Water market transaction	3,548.34	3,548.34 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Texas)	NO (Urban -municipal and industrial)	Water market transaction	845.78	845.78 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Utah)	NO (Urban -municipal and industrial)	Water market transaction	602.05	602.05 [b]	N/A	Reported	
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Washington)	NO (Urban -municipal and industrial)	Water market transaction	759.35	759.35 [b]	N/A	Reported	

Appendix 10. Municipal (USA) Water Right

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Brewer, J., Glennon, R. Ker, A. and Libecap, G. (7)	USA (Wyoming)	NO (Urban -municipal and industrial)	Water market transaction	2,617.63	2,617.63 [b]	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	NO (City of Edinburgh)	Water market transaction	1,015.85	1,015.85	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	NO (City of Edinburgh)	Water market transaction	1,177.24	1,177.24	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	NO (City of Edinburgh)	Water market transaction	981.03	981.03	N/A	Reported	
Chan, C. and Griffin, R.C. (14)	USA (Texas – Rio Grande valley)	NO (City of Edinburgh)	Water market transaction	1,079.14	1,079.14	N/A	Reported	

* Price elasticities noted in each paper were not necessarily used to derive the corresponding value estimates in all cases. [a] Ditch company shares. [b] Agriculture to urban sale price.

Appendix 11. Municipal (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Al-Ghuraiz, Y. and Enshassi, A. (71)	Palestinian Territory	YES	CVM	1,394.97	1,394.97	Mean	Reported	
Al-Ghuraiz, Y. and Enshassi, A. (71)	Palestinian Territory	YES	Price	492.34	492.34	Mean	Reported	
Anielski, M. and Wilson. S.J. (73)	Canada (Boreal region)	NO (Municipal)	Benefit transfer	64.20	64.20	Unclear	Reported	
Banda, B.M., Farolfi, S. and Hassan, R.M. (76)	South Africa (Steelport sub-basin)	YES	Demand function	1,365.81	1,365.81 [a]	Unclear	Reported	
Banda, B.M., Farolfi, S. and Hassan, R.M. (76)	South Africa (Steelport sub-basin)	YES	Demand function	2,036.98	2,036.98 [b]	Unclear	Reported	
Banda, B.M., Farolfi, S. and Hassan, R.M. (76)	South Africa (Steelport sub-basin)	YES	CVM	1,891.48	1,891.48 [c]	Unclear	Reported	
Banda, B.M., Farolfi, S. and Hassan, R.M. (76)	South Africa (Steelport sub-basin)	YES	CVM	2,886.50	2,886.50 [d]	Unclear	Reported	
Emerton, L (ed) (83)	Tanzania (Pagani River Basin)	YES	Price	5,684.01 – 7,105.02	6,394.51 [e]	Median	Summarised	
Emerton, L., Erdenesaikhan, N., De Veen, B., Tsogoo, D., Janchivdorj, L., Suvd, P., Enkhtsetseg, B., Gandolgor, G., Dorisuren, C., Sainbayar, D. and Enkhbaatar, A. (84)	Mongolia (Upper Tuul Valley)	YES	CVM	5,286.67	5,286.67	Unclear	Reported	
Gibbons, D.C. (26)	Canada (Ontario)	YES	Demand function	0 – 304.09	49.05 [f]	Median	Summarised	-0.75 [x] -1.07 [y]
Kanyoka, P., Farolfi, S. and Morardet, S. (91)	South Africa (Sekororo-Letsoalo area in the Limpopo Province)	YES	DCE	1,095.22	1,095.22 [g]	Unclear	Reported	
Kulshreshtha, S.N. (94)	Canada (Manitoba)	YES	Opportunity cost	867.32	867.32 [h]	N/A	Reported	-0.23 [z] -0.42 -0.53
Kulshreshtha, S.N. (94)	Canada (Manitoba)	YES	Opportunity cost	258.23	258.23 [i]	N/A	Reported	
Kulshreshtha, S.N. (94)	Canada (Manitoba)	NO (Municipal)	Opportunity cost	132.78	132.78 [j]	N/A	Reported	
Kulshreshtha, S.N. (94)	Canada (Manitoba)	YES	Demand curve	249.68 – 1,246.43	657.32 [h]	Median	Summarised	

Appendix 11. Municipal (Rest of the World)

Author (paper reference number) (1)	Country (location) (2)	Domestic specific? (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported /summarised (8)	Elasticity (where available) (9) *
Kulshreshtha, S.N. (94)	Canada (Manitoba)	YES	Demand curve	99.84 – 421	208.92 [i]	Median	Summarised	
Kulshreshtha, S.N. (94)	Canada (Manitoba)	NO (Municipal)	Demand curve	60.49 – 166.46	111.45 [j]	Median	Summarised	
Larson, B., Minten, B. and Razafindralambo, R. (97)	Madagascar (Province of Fianarantsoa)	YES	CVM	21,419.74	21,419.74	Median	Reported	
McCartney, M. P., Lankford, B. A., Mahoo, H. (102)	Tanzania (Usangu Plains)	YES	CVM	1,330.85	1,330.85	Mean	Reported	
Muller, R.A. (107)	Canada	NO (Municipal)	Demand curve	194.93	194.93 [k]	N/A	Reported	
Muller, R.A. (107)	Canada	NO (Municipal)	Demand curve	4,736.83	4,736.83 [l]	N/A	Reported	
Nieuwoudt, W.L., Backeberg, G.R. and Du Pleiss, H.M. (108)	South Africa	NO (Urban)	Benefit transfer	1,565.95	1,565.95	Unclear	Reported	-0.47
Nieuwoudt, W.L., Backeberg, G.R. and Du Pleiss, H.M. (108)	South Africa	NO (Urban)	Unclear	482.83	482.83	Unclear	Reported	
Perez-Pineda, F. and Quintanilla-Armijo, C. (110)	El Salvador	YES	Price	4,607.96	4,607.96	Mean	Reported	
Perez-Pineda, F. and Quintanilla-Armijo, C. (110)	El Salvador	YES	CVM	7,508.33	7,508.33	Mean	Reported	
Raje, D., Dhobe, P. and Deshpande, A. (113)	India (Mumbai)	YES	Price	148.07 – 222.10	185.08 [m]	Median	Summarised	
Raje, D., Dhobe, P. and Deshpande, A. (113)	India (Mumbai)	YES	Price	296.13 – 370.16	333.15 [m]	Median	Summarised	
Raje, D., Dhobe, P. and Deshpande, A. (113)	India (Mumbai)	YES	Price	592.26	592.26	N/A	Reported	
Rogers, P., Bhatia, R. and Huber, A. (117)	Thailand (Phuket)	YES	Price	2,212.92	2,212.92 [n]	Unclear	Reported	
Rogers, P., Bhatia, R. and Huber, A. (117)	Thailand (Phuket)	YES	Price	1,838.42	1,838.42 [o]	Unclear	Reported	
Rogers, P., Bhatia, R. and Huber, A. (117)	India (Jamshedpur, Subernarekha River Basin)	NO (Urban)	CVM	425.56	425.56 [p]	Mean	Reported	

Appendix 11. Municipal (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Rogers, P., Bhatia, R. and Huber, A. (117)	India (Jamshedpur, Subernarekha River Basin)	NO (Urban)	Unclear	794.95	794.95 [q]	Unclear	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Tegucigalpa)	YES	Price	180.32	180.32 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Tegucigalpa)	YES	Price	360.85	360.85 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Tegucigalpa)	YES	CVM	245.89	245.89 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (San Pedro Sula)	YES	Price	213.11	213.11 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (San Pedro Sula)	YES	Price	426.22	426.22 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (San Pedro Sula)	YES	CVM	213.11	213.11 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (San Pedro Sula)	YES	RP	803.26	803.26 [u]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Intermediate cities)	YES	Price	114.75	114.75 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Intermediate cities)	YES	Price	573.75	573.75 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Intermediate cities)	YES	CVM	163.93	163.93 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Honduras (Intermediate cities)	YES	RP	229.50	229.50 [u]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Nicaragua (Managua)	YES	Price	409.82	409.82 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Nicaragua (Managua)	YES	Price	770.47	770.47 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Nicaragua (Managua)	YES	CVM	262.29	262.29 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Nicaragua (Managua)	YES	RP	377.04	377.04 [u]	N/A	Reported	

Appendix 11. Municipal (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Sonsonate)	YES	Price	295.07	295.07 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Sonsonate)	YES	Price	295.07	295.07 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Sonsonate)	YES	CVM	524.58	524.58 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Sonsonate)	YES	RP	262.29	262.29 [u]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Santa Ana)	YES	Price	295.07	295.07 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Santa Ana)	YES	Price	278.68	278.68 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Santa Ana)	YES	CVM	508.18	508.18 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (Santa Ana)	YES	RP	311.47	311.47 [u]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (San Miguel)	YES	Price	295.07	295.07 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (San Miguel)	YES	Price	344.25	344.25 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (San Miguel)	YES	CVM	803.26	803.26 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	El Salvador (San Miguel)	YES	RP	278.68	278.68 [u]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Panama (Pananma City and Colon)	YES	Price	409.82	409.82 [r]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Panama (Pananma City and Colon)	YES	Price	1,163.90	1,163.90 [s]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Panama (Pananma City and Colon)	YES	CVM	836.04	836.04 [t]	N/A	Reported	
Walker, I., Ordonez, F., Serrano, P. and Halpern, J. (121)	Panama (Pananma City and Colon)	YES	RP	655.72	655.72 [u]	N/A	Reported	
Wang, H., Xie, J. and Li, H. (123)	China	YES	DCE	866.09 – 1,314.07	1,219.50 [v]	Median	Summarised	-0.355

Appendix 11. Municipal (Rest of the World)

Author (paper reference number) (1)	**Country (location) (2)**	**Domestic specific? (3)**	**Valuation approach (4)**	**Value range 2014 $/AF (5)**	**2014 $/AF (6)**	**Measure of central tendency (7)**	**Reported /summarised (8)**	**Elasticity (where available) (9) ***
Whittington, D., Lauria, D.T. and Mu, X. (124)	Nigeria (Onitsha)	NO (Municipal)	Price	983.98 – 42,639.02	22,959 [w]	Median	Summarised	

* Price elasticities noted in each paper were not necessarily used to derive the corresponding value estimates in all cases. [a] Consumer surplus per household for collective tap water. [b] Consumer surplus per household for river water. [c] CVM estimate – value of water for collective tap users. [d] CVM estimate – value of river water. [e] Median of recorded water prices set by local authorities. [f] Values refer to net consumer surplus i.e. the value of the water at source net of water utility costs. Values are derived for various reductions in the average monthly consumption. [g] Households without taps WTP for improvement in water service. [h] Value refers to groundwater use for domestic purposes by rural farms. [i] Value refers to non-farm groundwater use for domestic purposes. [j] Value refers to groundwater use for domestic purposes in the town of Neepawa. [k] Low estimate. [l] High estimate. [m] Median values in ranges given for domestic water users living in slums and chawls and multi stories residential buildings. [n] Value in use of urban consumers and hotels for vended water during summer months. [o] Full costs of water including environmental externalities. [p] Value in urban use of households. [q] Full costs of water including O&M costs, capital charges and environmental externalities. [r] Present water tariff. [s] Benchmark full-cost water tariff. [t] Contingent valuation estimate of the price at which consumption would be 30 m^3 (standard monthly benchmark consumption level). [u] Revealed preference of the price at which consumption would be 30 m^3 (standard monthly benchmark consumption level). [v] Median value across different income groups and scenarios presented. [w] Median price charged by water vendors for different small scale transactions. [x] Winter price elasticity. [y] Summer price elasticity. [z] Price elasticity of 0.23 refers to small communities and open areas; 0.42 refers to medium sized non-farm communities, and -0.53 refers to large rural non-farm communities.

Appendix 12. Waste Assimilation (USA)

Author (paper reference number) (1)	Country (location) (2)	Pollutant (3)	Point/non-point (4)	Valuation approach (5)	Value range 2014 $/AF (6)	2014 $/AF (7)	Measure of central tendency (8)	Reported/ Summarised (9)
Gibbons, D.C. (26)	USA (Colorado River)	Salinity	Point/non-point	Damages avoided	39.24	39.24	Mean	Reported
Gibbons, D.C. (26)	USA (Geographically non-specific)	Thermal	Point	Alternative cost	24.52	24.52	Unclear	Reported
Gray, S.L. and Young, R.A. (28)	USA (22 regions)	BOD	Point	Alternative cost	0.52-7.74	1.80 [a]	Median	Summarised
Gray, S.L. and Young, R.A. (28)	USA (22 regions)	BOD	Point	Alternative cost	1.13-16.51	3.83 [b]	Median	Summarised
Gray, S.L. and Young, R.A. (28)	USA (22 regions)	BOD	Point	Alternative cost	0.35-16.12	2.05 [c]	Median	Summarised
Gray, S.L. and Young, R.A. (28)	USA (22 regions)	BOD	Point	Alternative cost	0.17-7.56	0.96 [d]	Median	Summarised
Hastay, M. (33)	USA (Colombia River)	BOD	Point/non-point	Alternative cost	0.23	0.23	N/A	Reported
Meritt, L.B. and Mar, B.W. (50)	USA (Williamette River Basin)	BOD	Point	Alternative cost	2.85	2.85	Value of dilution water summed over the river reaches.	Reported
Meritt, L.B. and Mar, B.W. (50)	USA (Colombia River Basin)	BOD	Point	Alternative cost	0.21-4.46	1.03 [e]	Median	Summarised
Meritt, L.B. and Mar, B.W. (50)	USA (Colombia River Basin)	BOD	Point	Alternative cost	0.09-3.59	0.7 [f]	Median	Summarised
Meritt, L.B. and Mar, B.W. (50)	USA (Colombia River Basin)	BOD	Point	Alternative cost	0.07-2.54	0.52 [g]	Median	Summarised
Renshaw, E.F. (57)	USA (Geographically non-specific)	Non-specific	Point/non-point	Alternative cost	16.39	16.39	Mean	Reported
Russell, C.S. (58)	USA (Geographically non-specific)	Thermal	Point	Alternative cost	3.77	3.77	N/A	Reported

[a] Marginal value/minimum cost combination (3.75% discount rate and 50 year plant life). [b] Marginal value/minimum cost combination (6% discount rate and 30 year plant life). [c] Marginal value 70/50% treatment (6% discount rate and 30 year plant life). [d] Marginal value 70/50% treatment (6% discount rate and 30 year plant life). [e] Plant size 2.5mgd (millions of gallons per day). [f] Plant size 10 mgd (millions of gallons per day). [g] Plant size 50 mgd (millions of gallons per day).

Appendix 13. Waste water treatment

Author (paper reference number) (1)	**Country (location) (3)**	**Valuation type (4)**	**2014 $/AF (5)**	**Measure of central tendency (6)**	**Reported/ summarised (7)**
Hernández -Sancho, F and Sala-Garrido, R. (87)	Spain (Valencia)	Operating cost	1,531.22 [a]	Mean	Reported
Hernández -Sancho, F and Sala-Garrido, R. (87)	Spain (Valencia)	Operating cost	803.41 [b]	Mean	Reported
Hernández-Sancho, F., Molinos-Senante, M. and Sala-Garrido, R. (88)	Spain (Valencia)	Environmental benefit	1,601.26 [c]	N/A	Reported
Lavee, D. (99)	Israel (Lake Kinneret)	Damages avoided	362.44	N/A	Reported
Lavee, D. (99)	Israel (Lake Kinneret)	Operating cost (cost of filtration)	87.25	N/A	Reported
Lavee, D. (99)	Israel (Lake Kinneret)	WTP for filtration	251.69	N/A	Reported
Molinos-Senante, M., Hernández -Sancho, F and Sala-Garrido, R. (104)	Spain (Valencia)	Environmental benefit	2,061.89 [d]	Median	Summarised
Molinos-Senante, M., Hernández -Sancho, F and Sala-Garrido, R. (105)	Spain (Valencia)	Environmental benefit	610.22 [e]	Median	Summarised
Molinos-Senante, M., Hernández -Sancho, F and Sala-Garrido, R. (105)	Spain (Valencia)	Operating cost	418.84 [f]	Median	Summarised
Turpie, J., Day, E., Ross-Gillespie, R. and Louw, A. (120)	South Africa (South Western Cape Province)	Operating cost	240.93 [g]	Weighted mean	Reported

[a] Mean from plants in group A (224 WWTPs). [b] Mean from plants in group B (134 WWTPs). [c] Total environmental benefit per acre foot from removing five pollutant types at 43 WWTPs. [d] Median value across 13 WWTPs studied. [e] Median value across 22 WWTPs studied. [f] Median value across 22 WWTPs studied. [g] Weighted average across the 19 WWTPs studied.

Appendix 14. Wildlife Habitat (USA) Per Period

Author (paper reference number) (1)	Country (location) (3)	Wildlife type (4)	Valuation approach (5)	Value range 2014 $/AF (7)	2014 $/AF (8)	Measure of central tendency (9)	Reported/ summarised (10)
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (California – Central Valley)	Environmental purposes unspecified	Water market transaction	161.08	161.08 [a]	N/A	Reported
Bollman, F.H. (5)	USA (California - Toulumne River)	Salmon spawning	Unclear	98.09	98.09	N/A	Reported
Bush, A. (10)	USA (California – Trinity River)	Fish hatchery	Unclear	56.40	56.40	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Arizona)	Environmental purposes unspecified	Water market transaction	55.73	55.73	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (California)	Environmental purposes unspecified	Water market transaction	65.24	65.24	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Colorado)	Environmental purposes unspecified	Water market transaction	14.95	14.95	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Idaho)	Environmental purposes unspecified	Water market transaction	25.83	25.83	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Montana)	Environmental purposes unspecified	Water market transaction	2.72	2.72	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (New Mexico)	Environmental purposes unspecified	Water market transaction	8.84	8.84	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Oregon)	Environmental purposes unspecified	Water market transaction	154.95	154.95	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Washington)	Environmental purposes unspecified	Water market transaction	46.21	46.21	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	In-stream flow	Water market transaction	62.28	62.28	N/A	Reported

Appendix 14. Wildlife Habitat (USA) Per Period

Author (paper reference number) (1)	Country (location) (3)	Wildlife type (4)	Valuation approach (5)	Value range 2014 $/AF (7)	2014 $/AF (8)	Measure of central tendency (9)	Reported/ summarised (10)
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Threatened and endangered species	Water market transaction	79.27	79.27	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Fish	Water market transaction	55.48	55.48	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Wildlife	Water market transaction	48.29	48.29	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Wetlands	Water market transaction	74.87	74.87	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Ecosystem Services	Water market transaction	55.32	55.32	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (California)	Fish hatchery	Unclear	48.90	48.90	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (California)	Salmon spawning	Unclear	83.60	83.60	N/A	Reported
Moore, D. and Willey, Z. (51)	USA (California – Grasslands Water District)	Environmental purposes unspecified	Unclear	10.57 – 11.83	11.20 [b]	Median	Summarised
Postel, M. and Carpenter, S. (55)	USA (Colorado - Sacramento-San Joaquin Delta)	Migrating fish and wildlife refuges	Water market transaction	73.75	73.75	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA (Oregon – upper snake river)	Migrating salmon	Water market transaction	117.99	117.99	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA (California - San Luis Kesterson Wildlife Refuge)	Wetland maintenance	Water market transaction	30.84	30.84	N/A	Reported
Renshaw, E.F. (57)	USA (Geographically non-specific)	Commercial fishing	Value of catch	0.16	0.16	Mean	Reported

[a] Short-term lease value. [b] Median value within range given.

Appendix 15. Wildlife Habitat (USA) Water Right

Author (paper reference number) (1)	Country (location) (2)	Wildlife type (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Aylward, B., Seely, H., Hartwell, R. and Dengel, J. (2)	USA (Nevada – Truckee Basin)	Environmental purposes unspecified	Water market transaction	5,369.49	5,369.49	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Arizona)	Environmental purposes unspecified	Water market transaction	57.09	57.09	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (California)	Environmental purposes unspecified	Water market transaction	3,828.99	3,828.99	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Colorado)	Environmental purposes unspecified	Water market transaction	1,478.86	1,478.86	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Idaho)	Environmental purposes unspecified	Water market transaction	178.06	178.06	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Nebraska)	Environmental purposes unspecified	Water market transaction	1,079.24	1,079.24	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Nevada)	Environmental purposes unspecified	Water market transaction	1,352.45	1,352.45	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Oregon)	Environmental purposes unspecified	Water market transaction	330.30	330.30	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Utah)	Environmental purposes unspecified	Water market transaction	1,631.09	1,631.09	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Washington)	Environmental purposes unspecified	Water market transaction	1,128.17	1,128.17	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	In-stream flow	Water market transaction	1,023.51	1,023.51	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Threatened and endangered species	Water market transaction	1,372.84	1,372.84	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Fish	Water market transaction	694.57	694.57	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Wildlife	Water market transaction	1,385.07	1,385.07	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Wetlands	Water market transaction	1,510.12	1,510.12	N/A	Reported
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western states)	Ecosystem Services	Water market transaction	1,442.16	1,442.16	N/A	Reported

Appendix 15. Wildlife Habitat (USA) Water Right

Author (paper reference number) (1)	Country (location) (2)	Wildlife type (3)	Valuation approach (4)	Value range 2014 $/AF (5)	2014 $/AF (6)	Measure of central tendency (7)	Reported/ summarised (8)
Moore, D. and Willey, Z. (51)	USA (Nevada – Stillwater Wildlife Refuge)	Environmental purposes unspecified	Unclear	473.19	473.19 [a]	Unclear	Reported
Postel, M. and Carpenter, S. (55)	USA (Nevada – Lohonton Valley Wetlands)	Wetland maintenance	Water market transaction	338.02 – 507.03	422.52 [b]	Median	Summarised

[a] This appears to be a capitalised asset value given its size. However, it is not explicitly noted as such. [b] Median value within range given.

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Flow variation (3)**	**Recreation activity (4)**	**Site characteristics (5)**	**Valuation approach (6)**	**Value range 2014 $/AF (7)**	**2014 $/AF (8)**	**Measure of central tendency (9)**	**Reported/ summarised (10)**
Amirfathi, P., Narayanan, R. and Bishop, B. and Larson, D. (1)	USA (Utah - Blacksmith River)	25% of peak 1982 levels	Fishing	River	TCM	140.69	140.69	N/A	Reported
Amirfathi, P., Narayanan, R. and Bishop, B. and Larson, D. (1)	USA (Utah - Little Bear River)	25% of peak 1982 levels	Fishing	River	TCM	58.86	58.86	N/A	Reported
Bishop, R., Boyle, K., Welsh, M., Baumgartner, R. and Rathbun, P. (4)	USA (Arizona – Colorado River)	Low flow 10,000 CFS	Rafting	River	CVM	1.82	1.82	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Southwest)	Value of additional flows during ‘low flow’ periods.	Anglers	River	Unspecified	33.23	33.23	N/A	Reported
Booker, J.F. and Colby, B.G. (6)	USA (Southwest)	Value of additional flows during ‘low flow’ periods.	Shoreline recreation	River	Unspecified	8.67	8.67	N/A	Reported
Colby, B.G. (15)	USA (Colorado)	Low flows	Fishing	Unspecified	Unspecified	35.49	35.49	N/A	Reported
Cooper, J. and Loomis, J.B. (17)	USA (California - Kesterson Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	1.48 – 6.26	3.75 [a]	Median	Summarised.
Cooper, J. and Loomis, J.B. (17)	USA (California – Los Banos Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	3.83 – 20.63	15.03 [a]	Median	Summarised.
Cooper, J. and Loomis, J.B. (17)	USA (California - Mendota Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	9.05 – 38.27	29.73 [a]	Median	Summarised.

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Flow variation (3)**	**Recreation activity (4)**	**Site characteristics (5)**	**Valuation approach (6)**	**Value range 2014 $/AF (7)**	**2014 $/AF (8)**	**Measure of central tendency (9)**	**Reported/ summarised (10)**
Cooper, J. and Loomis, J.B. (17)	USA (California – San Luis Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	2.85 – 13.33	10.58 [a]	Median	Summarised.
Cooper, J. and Loomis, J.B. (17)	USA (California - Volta Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	3.42 – 14.44	8.64 [a]	Median	Summarised.
Cooper, J. and Loomis, J.B. (17)	USA (California - Merced Wildlife Refuge)	An additional acre foot	Waterfowl hunting	Wetland	TCM (Zonal)	0.41 – 13.49	1.68 [a]	Median	Summarised.
Creel, M. and Loomis, J. (18)	USA (California - San Joaquin Valley)	62,880 AF	Wildlife viewing, fishing and waterfowl hunting	Six river destinations, National Wildlife Refuges and State Wildlife Management Areas	Linked selection model and count data trip frequency model	512.10 – 588.15	550.12 [b]	Median	Summarised
Daubert, J.T. and Young, R.A. (20)	USA (Colorado – Poudre River)	100 – 700 CFS	Fishing	River	CVM	-37.07 – 59.07	4.15 [c]	Median	Summarised
Duabert, J.T. and Young, R.A. (21)	USA (Colorado – Poudre River)	50 – 700 CFS	Fishing	River	CVM	-37.07 – 92.67	9.77 [d]	Median	Summarised
Duffield, J.W., Neher, C.J. and Brown, T.C. (22)	USA (Montana – Big Hole River)	100 – 2,000 CFS	Predominantly fishing	River	CVM (Discrete Choice)	1.95 – 44.69	21.46 [e]	Median	Summarised
Duffield, J.W., Neher, C.J. and Brown, T.C. (22)	USA (Montana – Bitterroot River)	100 – 2,000 CFS	Predominantly fishing	River	CVM (Discrete Choice)	-0.84 – 18.10	9.62 [e]	Median	Summarised
Gibbons, D.C. (26)	USA (Washington – Yakima River System)	Minimum flows of 805 CFS	Fishing	River	Unspecified	45.17	45.17	N/A	Reported
Hansen, L.T. and Hallam, A. (29)	USA (multiple ASAs)	Unspecified	Fishing (Trout)	River	Unclear	0.29 – 4,073.21	8.45 [f]	Median	Summarised
Hansen, L.T. and Hallam, A. (29)	USA (multiple ASAs)	Unspecified	Fishing (Bass)	River	Unclear	0.25 – 4,849.23	7.66 [f]	Median	Summarised

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Flow variation (3)**	**Recreation activity (4)**	**Site characteristics (5)**	**Valuation approach (6)**	**Value range 2014 $/AF (7)**	**2014 $/AF (8)**	**Measure of central tendency (9)**	**Reported/ summarised (10)**
Harpman, D.A. (30)	USA (Colorado – Taylor River)	Critical winter low flow 40 CFS	Fishing	River	CVM	3.26	3.26	N/A	Reported
Johnson, N.S. and Adams, R.M. (37)	USA (Oregon – John Day River)	204 (mean summer flow) – 2,700 (mean spring flow)	Fishing	River	CVM	-0.58 – 4.29	0.33 [g]	Median	Summarised
Lansford Jr, N.H. and Jones J.L. (44)	USA (Texas - Highland Lakes chain)	N/A	Unspecified	Lake	HPM	6.04 – 64.21	17.93 [h]	Median	Summarised
Loomis, J. and McTernan, J. (46)	USA (Colorado – Poudre River)	500 – 2,300 CFS	Non-commercial kayakers and river rafters	River	CVM	-156.97 – 250.41	139.77 [i]	Median	Summarised
Loomis, J. and McTernan, J. (46)	USA (Colorado – Poudre River)	500 – 2,300 CFS	Non-commercial kayakers and river rafters	River	TCM	-141.92 – 235.46	149.98 [i]	Median	Summarised
Loomis, J.B. (47)	USA (Colorado - urban river in City of Fort Collins)	Annual value of flow (unknown)	Numerous/not fully specified	River	CVM	72.67	72.67	Unspecified	Reported
Loomis, J.B. and Creel, M. (48)	USA (California - San Joaquin River)	62,800 AF	Fishing, waterfowl hunting and wildlife viewing	River	TCM - linked site choice and trip frequency models	69.74 – 179.55	135.13 [j]	Median	Summarised
Loomis, J.B. and Creel, M. (48)	USA (California - Stanislaus River)	10,000 AF	Fishing, waterfowl hunting and wildlife viewing	River	TCM - linked site choice and trip frequency models	16.70 – 20.74	19.86 [j]	Median	Summarised
Loomis, J.B., Quattlebaum, K., Brown, T.C. and Alexander, S.J. (49)	USA (Western USA)	N/A	Unspecified	Unspecified	Water market transaction (lease)	13.32	13.32	Mean	Reported
Moore, D. and Willey, Z. (51)	USA (New Mexico)	Unspecified	Unspecified	Unspecified	Unspecified	25.24 – 42.59	33.91 [k]	Median	Summarised

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Flow variation (3)**	**Recreation activity (4)**	**Site characteristics (5)**	**Valuation approach (6)**	**Value range 2014 $/AF (7)**	**2014 $/AF (8)**	**Measure of central tendency (9)**	**Reported/ summarised (10)**
Narayanan, R. (53)	USA (Utah – Blacksmith Fork River)	Low flow 80 CFS	Camping, hiking and fishing	River	TCM	1.86	1.86	N/A	Reported
Neher, C.J. (54)	USA (Montana – Bitterroot River)	25% decline in flows - total discharge 479,080 AF.	Fishing	River	TCM	12.43	12.43	N/A	Reported
Neher, C.J. (54)	USA (Montana – Upper Clark Fork)	25% decline in flows - total discharge 1,700,970 AF.	Fishing	River	TCM	1.16	1.16	N/A	Reported
Neher, C.J. (54)	USA (Montana – Upper Flathead)	25% decline in flows - total discharge 7,251,400 AF.	Fishing	River	TCM	0.42	0.42	N/A	Reported
Neher, C.J. (54)	USA (Montana – Upper Yellowstone)	25% decline in flows - total discharge 2,163,910 AF.	Fishing	River	TCM	9.68	9.68	N/A	Reported
Neher, C.J. (54)	USA (Regional 19 river model)	25% decline in flows - total discharge 45,727,381 AF.	Fishing	River	TCM	1.96	1.96	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA Colorado	Leaving water in high mountain reservoirs for an additional two weeks in August.	Reservoir recreation	Reservoir	Unspecified	75.71	75.71	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA (Northern Utah)	Additional AF when flows were 20-25% of peak levels.	River recreation	River	Unspecified	126.18	126.18	N/A	Reported

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	**Country (location) (2)**	**Flow variation (3)**	**Recreation activity (4)**	**Site characteristics (5)**	**Valuation approach (6)**	**Value range 2014 $/AF (7)**	**2014 $/AF (8)**	**Measure of central tendency (9)**	**Reported/ summarised (10)**
Postel, M. and Carpenter, S. (55)	USA Colorado	Additional AF above the 35% flow level.	Fishing - mountain stream	River	Unspecified	33.12	33.12	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA Colorado	Additional AF above the 35% flow level.	Kayaking	Unspecified	Unspecified	7.89	7.89	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA Colorado	Additional AF above the 35% flow level.	Rafting	Unspecified	Unspecified	6.31	6.31	N/A	Reported
Postel, M. and Carpenter, S. (55)	USA Colorado	Low flows	Shoreline recreation	Unspecified	Unspecified	23.66	23.66	N/A	Reported
Walsh, R.G., Auckerman, R. and Milton, R. (62)	USA (Colorado – Front Range)	Leaving water in reservoirs for an additional 16.7 days in August.	Unspecified	Reservoir	CVM	86.83	86.83	N/A	Reported
Walsh, R.G., Ericson, R., Arostegy, D. and Hansen, M. (68)	USA (Colorado - West slop Rocky Mountains)	0 – 100% of maximum flow	Fishing	River	CVM	-29.44 – 26.16	2.36 [l]	Median	Summarised
Walsh, R.G., Ericson, R., Arostegy, D. and Hansen, M. (63)	USA (Colorado - West slop Rocky Mountains)	0 – 100% of maximum flow	Kayaking	River	CVM	0.13 – 7.44	1.71 [l]	Median	Summarised
Walsh, R.G., Ericson, R., Arostegy, D. and Hansen, M. (63)	USA (Colorado - West slop Rocky Mountains)	0 – 100% of maximum flow	Rafting	River	CVM	0.02 – 4.8	0.91 [l]	Median	Summarised
Ward, F. (64)	USA (New Mexico)	Water in the stream in late summer.	Angling and boating	River	TCM	51.33	51.33	N/A	Reported
Ward, F.A. (65)	USA (New Mexico - Rio Chama River)	Optimal release	Fishing and boating	River	TCM	28.95 – 50.33	49.44 [m]	Median	Summarised

Appendix 16. Recreation (USA)

Author (paper reference number) (1)	Country (location) (2)	Flow variation (3)	Recreation activity (4)	Site characteristics (5)	Valuation approach (6)	Value range 2014 $/AF (7)	2014 $/AF (8)	Measure of central tendency (9)	Reported/ summarised (10)
Ward, F.A., Roach, B.A. and Henderson, J.E. (66)	USA (California – Sacramento)	Differing reservoir fill levels (40-100%)	Numerous/not fully specified	Reservoir	TCM	8.64 – 1,022	51.42 [n]	Median	Summarised

[a] Median value across the different regression models and the lower, upper and average value estimates that each model generated. [b] Median value across two water redistribution scenarios. [c] Median value across different rates of flow and across different months (May to October). [d] Median value across different rates of flow, months and TCM models. [e] Median value across different rates of flow in CFS. [f] Median value across multiple ASAs. [g] Median value across different rates of flow in different seasons. [h] Median value across different discount rates and discounting periods. [i] Median value across different rates of flow in CFS. [j] Median value across different flow release schedules. [k] Median value within range given. [l] Median values across different maximum flows and lengths of river. [m] Median across high and low run off years and the 1982 season. [n] Median value across different reservoir fill levels.

Appendix 17 – Non-normality of dependent and independent variables used in regression analysis (Chapter Three Part Three).

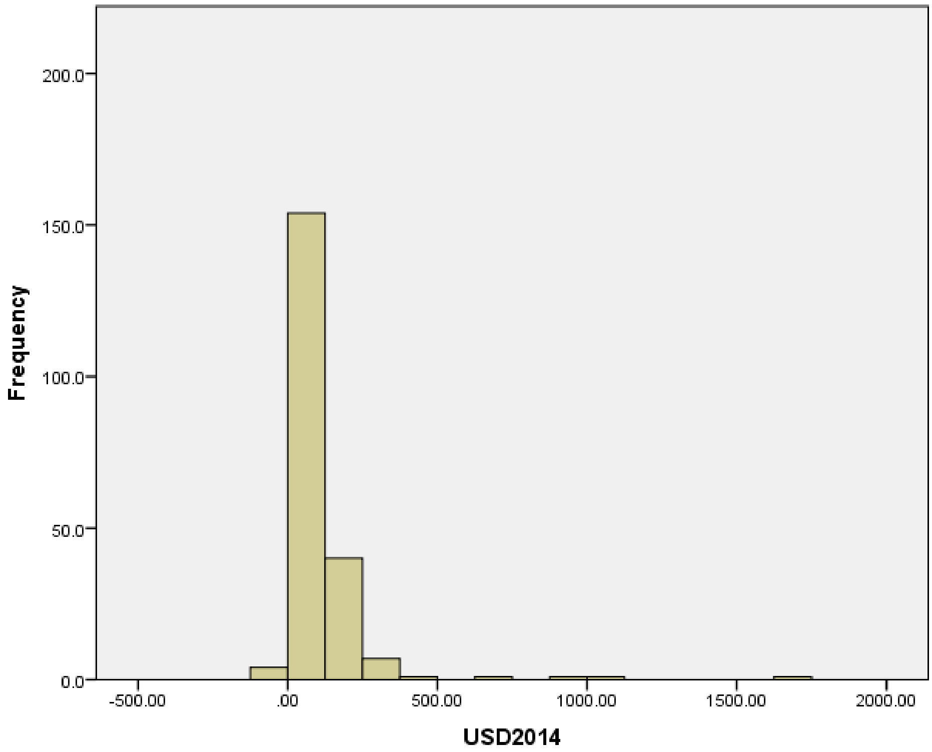


Figure 3.24 Frequency distribution for dependent variable (USD 2014).

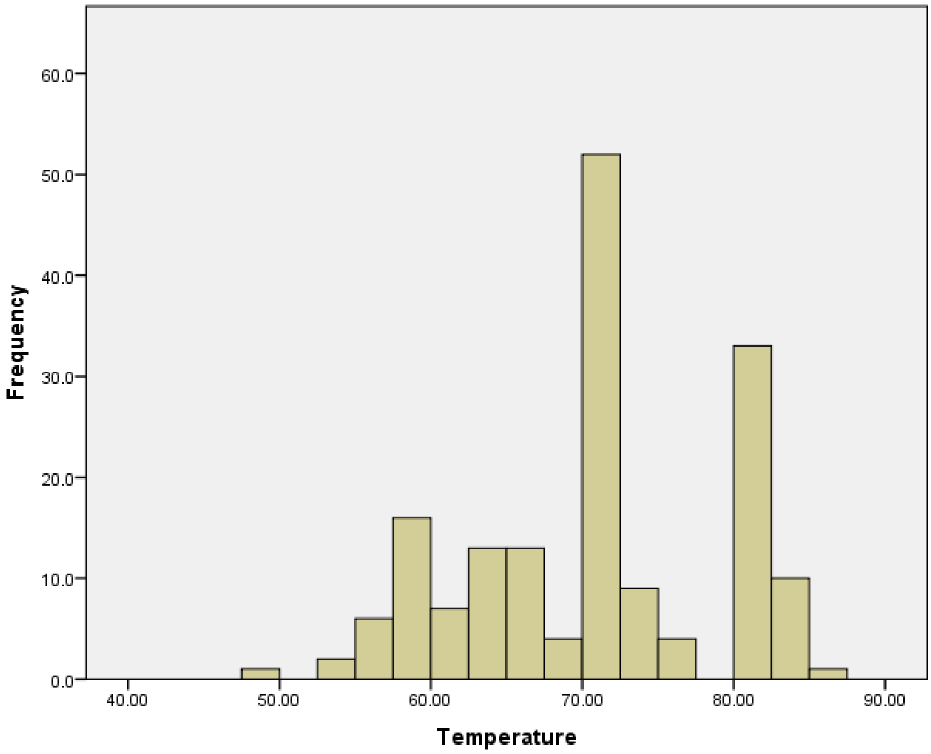


Figure 3.25 Frequency distribution for independent variable (temperature).

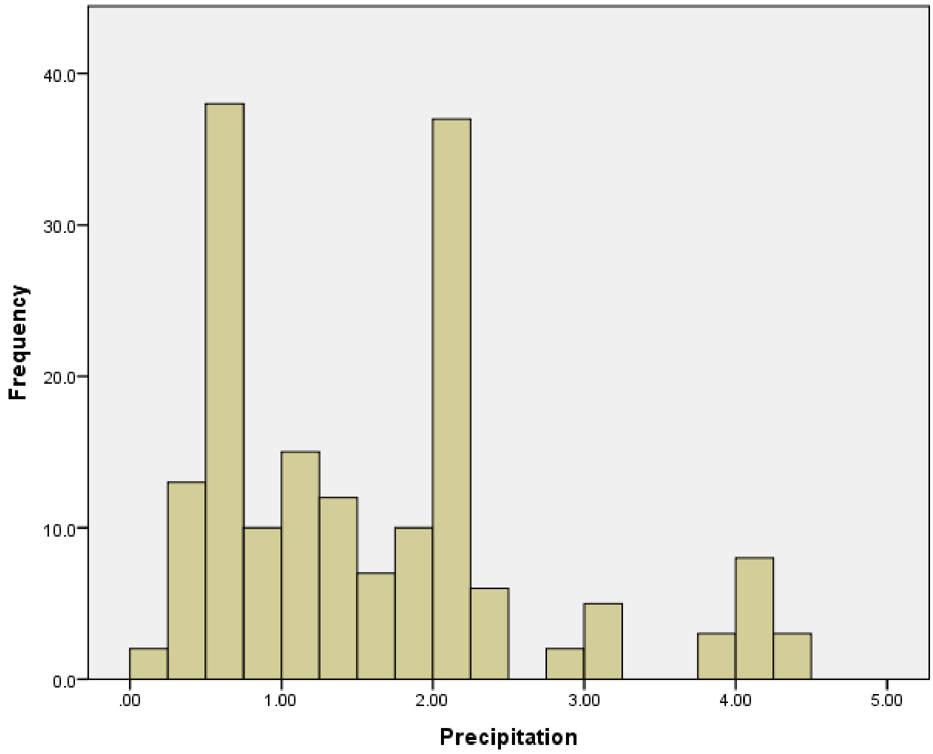


Figure 3.26 Frequency distribution for independent variable (precipitation).

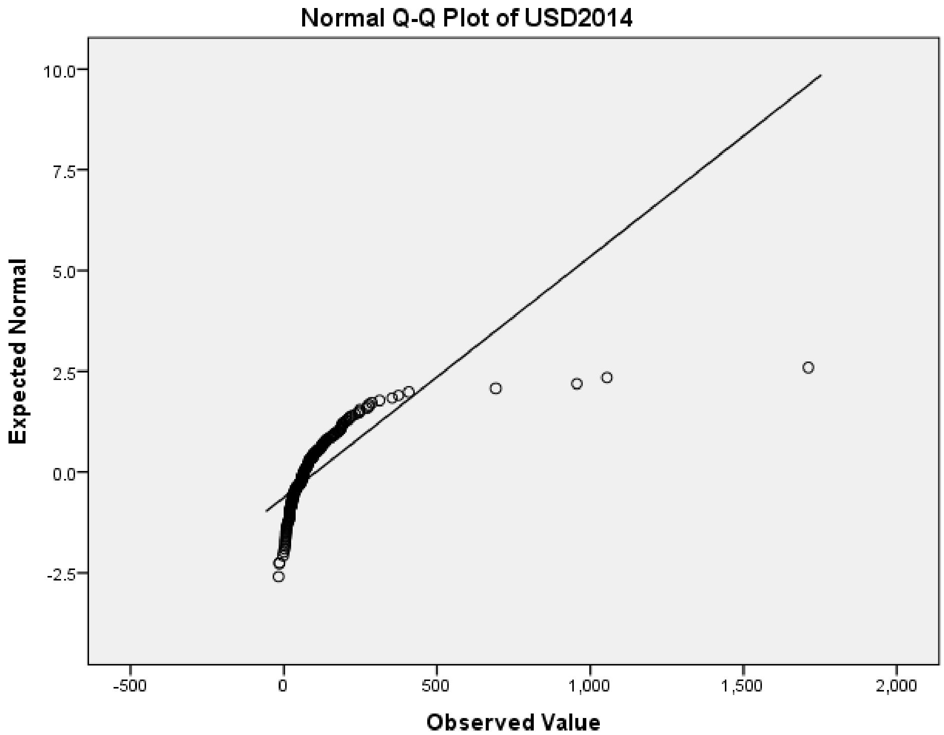


Figure 3.27 Normal Q-Q Plot of dependent variable (USD 2014).

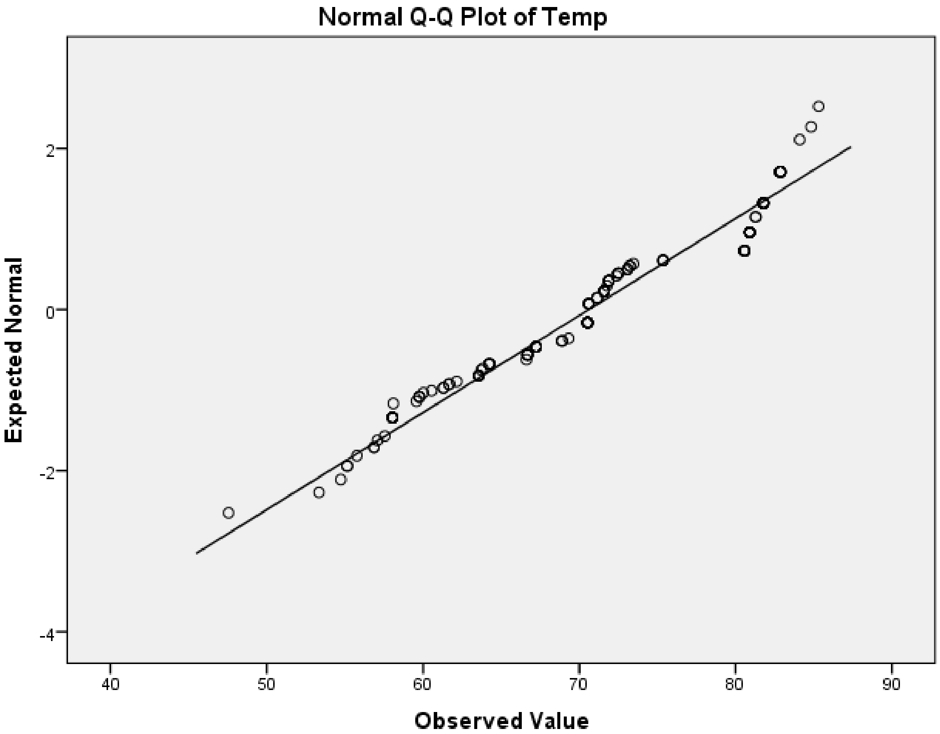


Figure 3.28 Normal Q-Q Plot of independent variable (Temperature).

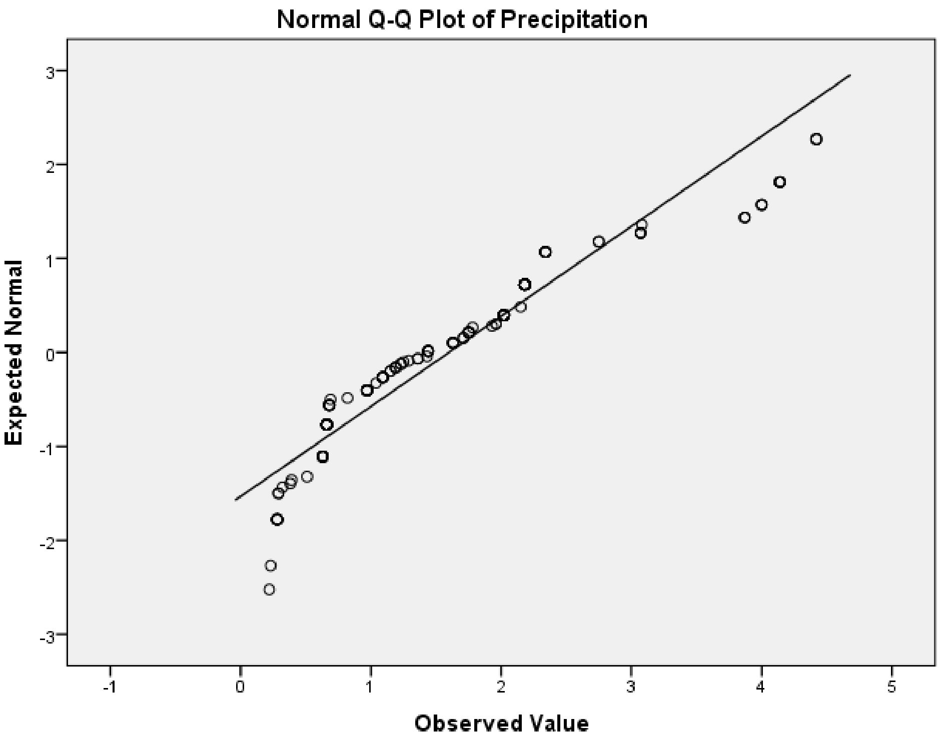


Figure 3.29 Normal Q-Q Plot of independent variable (Precipitation).

Appendix 18 – Crop parameters used in CROPWAT model (FAO, 2015b)

Table A. Crop development stages

Init. (L_{ini})	Dev. (L_{dev})	Mid (L_{mid})	Late (L_{late})	Total	Source
45	30	70	20	165	Allen et al. (1998)

Notes: Crop development stage values were amended in the CROPWAT model to reflect a late harvest potato crop that stays in the ground for approximately 165 days (Allen et al. 1998).

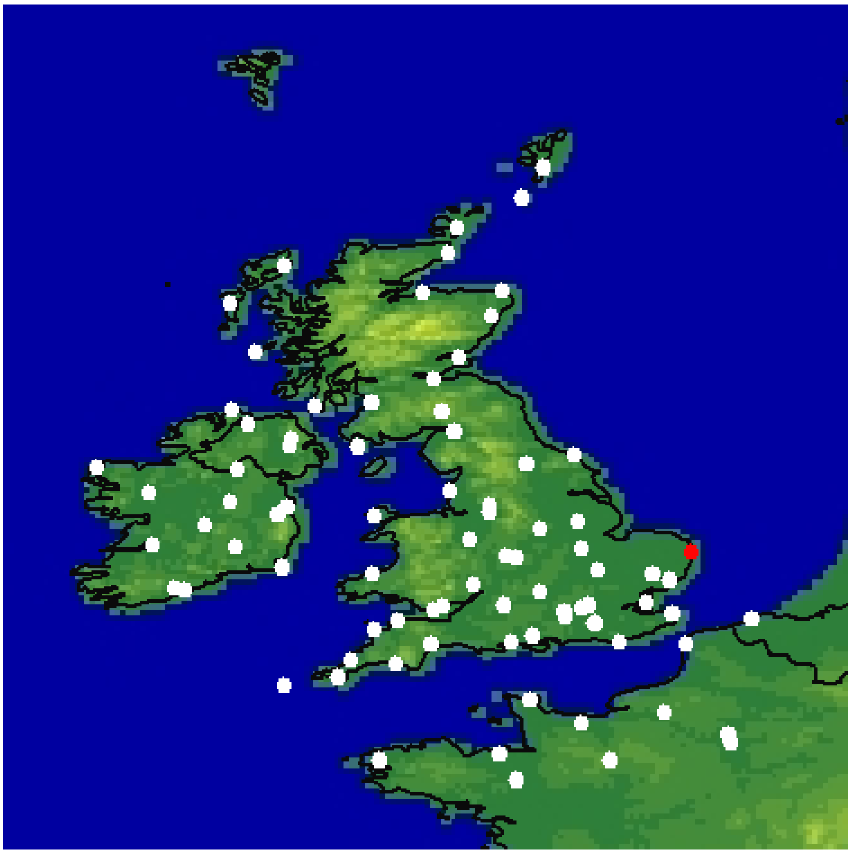
Table B. Crop coefficients as populated by CROPWAT based on potato crop profile

Kc_{ini}	Kc_{mid}	Kc_{end}	Maximum Crop Height (m)	Source
0.50	1.15	0.75	0.6	CROPWAT model

Table C. Maximum effective rooting depth, yield response factor and critical depletion fractions as populated by CROPWAT based on potato crop profile

Critical depletion fraction	Yield response function	Maximum effective rooting depth	Source
0.25-0.50	1.10	0.60	CROPWAT model

Appendix 19 – Gorleston meteorological station (FAO, 2015a)



Appendix 20 – Raw rainfall data used in CROPWAT model.

Rainfall in mm	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	Average
Jan	7.5	90.5	74	43	63	54	41	61.5	90.5	69.5	59.5
Feb	56.5	38.5	23	64	113	50	51.5	78	77.5	34.5	58.7
Mar	39	46.5	101	33.5	42	10	57	48.5	35	48.5	46.1
Apr	42.5	0	74	16.5	16.5	7	105.5	20.5	22	25	33.0
May	57	148.5	22	41.5	50.5	12	54.5	54.5	136	80	65.7
Jun	26	133.5	46.5	60	63.5	86.5	95	24.5	32.5	25	59.3
Jul	21.5	91	52.5	111.5	60	56.6	117	18	74	154	75.6
Aug	188	104.5	142.5	15	134	78.5	86.5	33	76.5	81	94.0
Sep	83.5	50	78	23	73.5	25.5	48.5	62	15	83.5	54.3
Oct	61	55.5	103.5	54.5	74.5	37	87	104.5	91	63.5	73.2
Nov	89	66	103.5	118.5	106	20.5	75	59	99.5	102	83.9
Dec	46	49	67.5	137	32	67	97	54.5	64	56.5	67.1
Total	717.5	873.5	888	718	828.5	504.6	915.5	618.5	813.5	823	770.1

Appendix 21 – Rainfall data processing method and stage-by-stage results (see FAO, 2008, p.7-8).

- Stage 1 – arrange rainfall data in descending order of magnitude and tabulate plotting position according to the following formula:

  Fa = 100* m/ (n+1)

  Where n = number of records

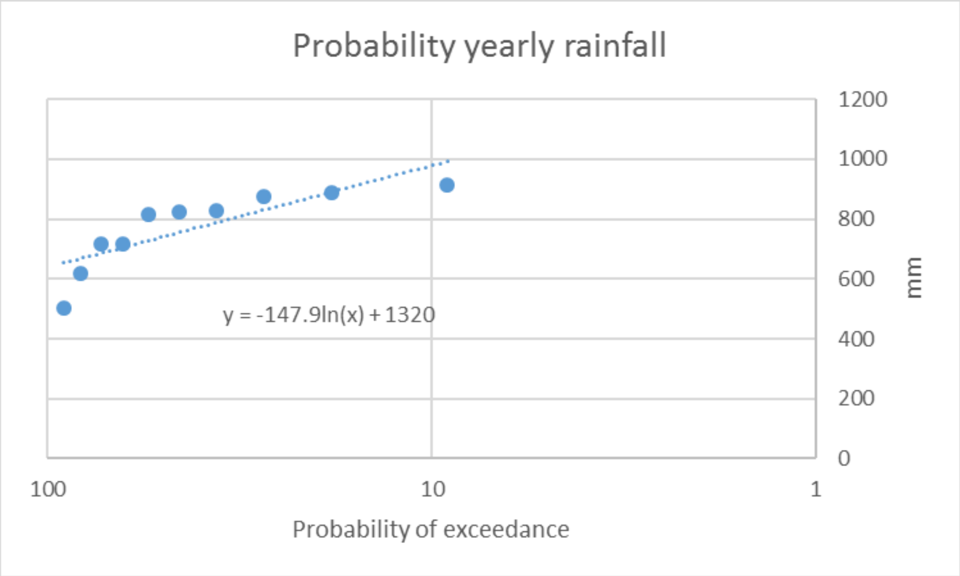
  m = rank number

  Fa = plotting position

Year	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Rain	717.5	873.5	888	718	828.5	504.6	915.5	618.5	813.5	823
Rank No.	8	3	2	7	4	10	1	9	6	5

Rank No.	1	2	3	4	5	6	7	8	9	10
Rain	915.5	888	873.5	828.5	823	813.5	718	717.5	618.5	504.6
Fa %	9.1	18.2	27.3	36.4	45.5	54.5	63.6	72.7	81.8	90.9

- Stage 2 – plot values on a log normal scale and obtain the logarithmic regression equation.



- Stage 3 – calculate year values at 20%, 50% and 80% probability.
- Stage 4 – for a dry year, calculate monthly values using the following equation:

$$P_{idry} = P_{iav} * \frac{P_{dry}}{P_{av}}$$

Where P_{iav} = average monthly rainfall for month 1

P^{idry} = monthly rainfall dry year for month 1

P_{av} = average yearly rainfall

P_{dry} = yearly rainfall at 80% probability of exceedance.

Value for wet and normal years can be calculated in the same way. Results for rainfall at Farm 1 can be seen in the table below:

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Average	59.45	58.65	46.1	32.95	65.65	59.3	75.61	93.95	54.25	73.2	83.9	67.05	770.06
Dry	51.9	51.2	40.2	28.7	57.3	51.7	66.0	82.0	47.3	63.9	73.2	58.5	671.9
Wet	67.7	66.8	52.5	37.5	74.8	67.5	86.1	107.0	61.8	83.4	95.5	76.4	876.9
Normal	57.2	56.5	44.4	31.7	63.2	57.1	72.8	90.5	52.2	70.5	80.8	64.6	741.4

Appendix 22 – Pre-populated soil parameters for medium (loam) soil (FAO, 2015b).

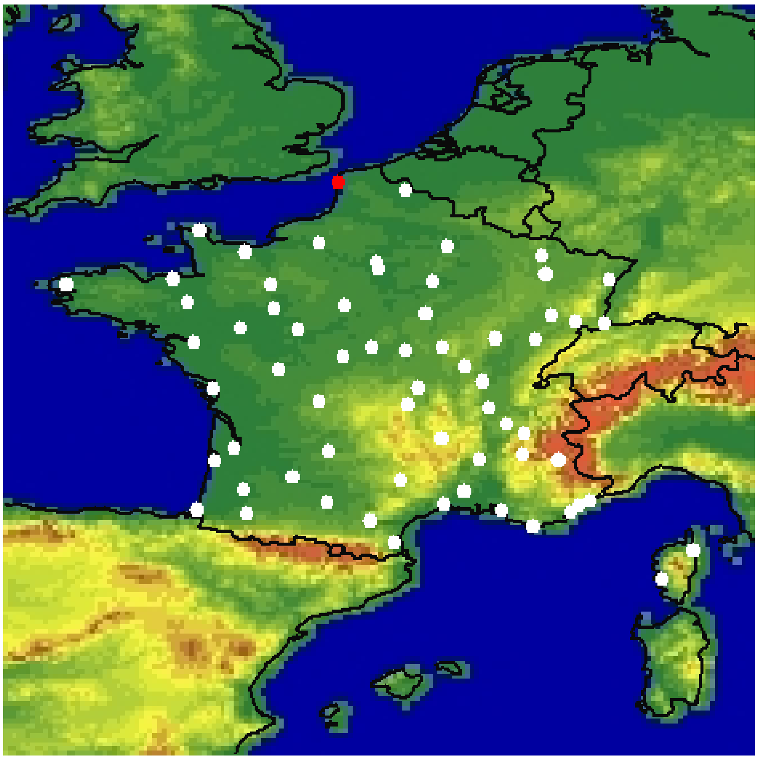
General data associated with medium (loam) soil as populated by CROPWAT

Soil Parameter	Soil data	Source
Total available soil moisture (FC-WP)	290 mm/meter	CROPWAT model
Maximum infiltration rate	40 mm/day	CROPWAT model
Maximum rooting depth	900 centimetres	CROPWAT model
Initial soil moisture depletion (as % TAM)	0 %	CROPWAT model
Initial available soil moisture	290 mm/meter	CROPWAT model

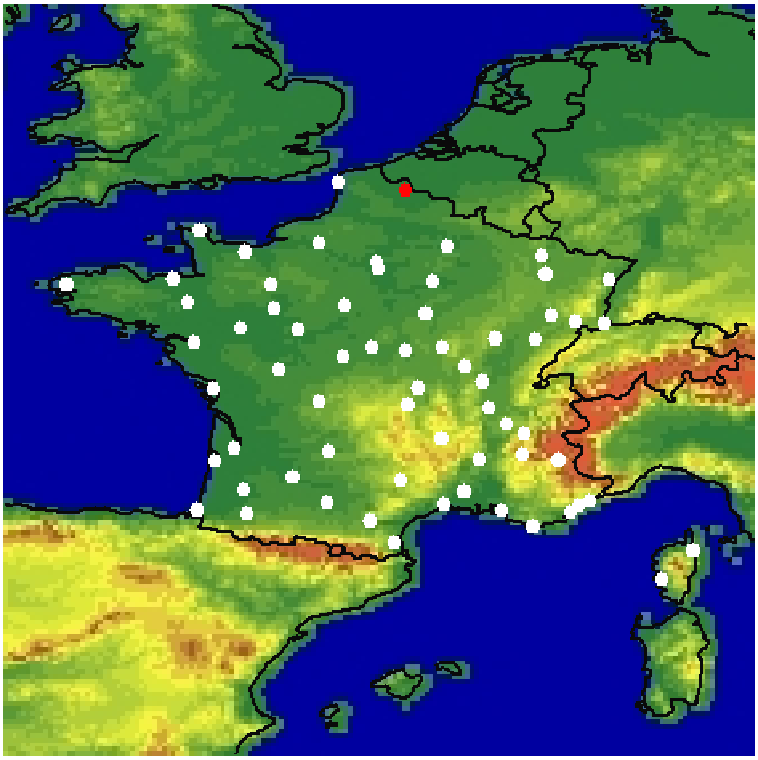
Appendix 23 – Example output from CROPWAT using the CWR option (Farm 1, normal year)

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.	ET green mm/period	ET blue mm/period
			coeff	mm/day	mm/dec	mm/dec	mm/dec		
Apr	1	Init	0.5	1.13	1.1	1	1.1	1	0.1
Apr	2	Init	0.5	1.23	12.3	8.3	4	8.3	4
Apr	3	Init	0.5	1.37	13.7	11.8	1.9	11.8	1.9
May	1	Init	0.5	1.52	15.2	16.9	0	15.2	0
May	2	Init	0.5	1.66	16.6	20.4	0	16.6	0
May	3	Deve	0.56	2	22	19.4	2.6	19.4	2.6
Jun	1	Deve	0.81	3.06	30.6	17.3	13.3	17.3	13.3
Jun	2	Deve	1.06	4.24	42.4	16.5	25.9	16.5	25.9
Jun	3	Mid	1.24	4.93	49.3	18.1	31.1	18.1	31.2
Jul	1	Mid	1.25	4.92	49.2	20.1	29.2	20.1	29.1
Jul	2	Mid	1.25	4.89	48.9	21.4	27.5	21.4	27.5
Jul	3	Mid	1.25	4.85	53.4	22.9	30.5	22.9	30.5
Aug	1	Mid	1.25	4.89	48.9	25.7	23.3	25.7	23.2
Aug	2	Mid	1.25	4.89	48.9	27.8	21.1	27.8	21.1
Aug	3	Mid	1.25	4.32	47.5	23.8	23.6	23.8	23.7
Sep	1	Late	1.15	3.42	34.2	17.9	16.4	17.9	16.3
Sep	2	Late	0.95	2.41	24.1	13.9	10.2	13.9	10.2
Sep	3	Late	0.84	1.84	1.8	1.6	1.8	1.6	0.2
					560.1	304.8	263.5	299.3	260.8
					Conversion to m3		10	2993	2608
					Yield ton/ha		49	61	53

Appendix 24 – Boulogne and Lille meteorological stations (FAO, 2015a)



Boulogne meteorological station



Lille meteorological station

Appendix 25 – Climate data comparison (Gorleston, Boulogne and Lille) (FAO, 2015a)

Gorleston UK	Min Temp °C	Max Temp °C	Humidity %	Wind km/day	Sun hours	Rad MJ/m²/day	ETo mm/day
January	-3.7	11.7	89	501	1.8	2.8	0.98
February	-2.8	12.1	88	475	2.3	4.8	1.12
March	-2	14.9	83	501	4.2	9.1	1.85
April	0.7	17.7	84	475	5.5	13.8	2.46
May	2	20.1	80	441	7	18.1	3.32
June	2.8	23.8	80	406	7.2	19.4	4
July	6.4	25.1	81	397	7	18.6	3.93
August	8.5	24.7	74	397	6.3	15.6	3.83
September	8.9	23.1	85	432	5.1	11	2.54
October	6.2	19	88	432	3.7	6.5	1.52
November	1.8	14.3	88	493	1.9	3.2	1.07
December	-0.3	12.3	89	501	1.5	2.2	0.87
Average	**2.4**	**18.2**	**84**	**454**	**4.5**	**10.4**	**2.29**

Boulogne France	Min Temp °C	Max Temp °C	Humidity %	Wind km/day	Sun hours	Rad MJ/m²/day	ETo mm/day
January	1.9	6	87	380	0.8	2.6	0.57
February	2	6.2	85	346	1.5	4.6	0.73
March	3.3	9.2	82	337	2.8	8.2	1.2
April	6.1	11.8	79	346	4.6	13.1	1.9
May	8.7	15.2	80	311	4.8	15.6	2.42
June	11.6	17.7	82	285	5.3	17.1	2.77
July	13.9	19.7	83	294	4.8	16.1	2.83
August	14.3	20	84	277	4.4	13.7	2.5
September	12.8	18.4	83	285	3.7	10.1	1.96
October	9.7	14.3	83	277	2.6	6.1	1.26
November	5.7	10	85	320	0.7	2.8	0.81
December	3.2	7.2	87	337	0.3	1.9	0.58
Average	**7.8**	**13**	**83**	**316**	**3**	**9.3**	**1.63**

Lille France	Min Temp °C	Max Temp °C	Humidity %	Wind km/day	Sun hours	Rad MJ/m²/day	ETo mm/day
January	0.1	5	87	337	0.8	2.6	0.52
February	0.2	5.9	85	302	1.5	4.6	0.68
March	2.3	10.1	81	294	2.8	8.2	1.23
April	4.6	13.7	77	302	4.5	13.1	2.06
May	7.5	17.4	75	268	4.8	15.6	2.75
June	10.2	20.5	78	242	5.3	17.1	3.13
July	12.2	22.2	78	251	4.8	16.1	3.21
August	12.4	22.4	80	233	4.4	13.7	2.8
September	10.5	19.6	81	242	3.7	10.1	2.08
October	7	14.4	86	233	2.6	6.1	1.12
November	3.7	9	89	277	0.6	2.8	0.62
December	1.1	5.8	90	294	0.3	2	0.43
Average	**6**	**13.8**	**82**	**273**	**3**	**9.3**	**1.72**