Optimisation and valuation of water use in Scotland

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Declaration

The candidate confirms that the work submitted is her own, except where work which has formed part of jointly-authored publications has been included. The thesis contains five chapters intended for publication in peer-reviewed journals. Details of each proposed publication, including an outline of the candidate and co-authors contributions, are stated at the start of the relevant chapters. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others. No cited part of this work has been submitted for any other degree or professional qualification.

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Lay Summary

Water has a variety of uses some of which are mediated by water quality, and some of which are mutually exclusive. How should any country rationally consider the allocation of water resources when they become scarce? In a resource abundant country, this might not always be a relevant question, but even in Scotland we can find examples of competition for uses and these competing demands may be exacerbated by climate change. Irrespective of climate scenarios, a country should seek to maximise the "wellbeing" return to its natural assets including water resources. This thesis explores how this might be done as a part of the Hydro Nation agenda which looks to maximise the value of water resources in Scotland.

The literature that considers the economic value of water recognises two related attributes that need to be revealed to optimise the value of water to any society. The first is the total valuation of water uses, the second relates to the ability to trade water rights.

Estimating the value and current allocation of water is the initial step to derive mechanisms that would shift water to high value uses. This agenda is becoming more central to debates about water regulation, including the Water Framework Directive, and as water demands increasingly compete due to climate change. This thesis explores the valuation of different water uses in Scotland and the current allocation of water between them to contribute to the discussion of moving towards an allocation that maximises social value with reference to relevant measures and policies. In doing so, initially the positioning of the work in the wider environmental and ecological economics literatures is discussed. There is also discussion of how the work relates to the overall conceptualisation of "water as an economic good"

In this thesis a portfolio of water uses is constructed to identify how much water is allocated to which use and at which value. The results show that, not surprisingly, the highest water allocation and value creation from water use in Scotland is in the hydropower sector. However, the higher unit values are created in consumptive uses; the highest unit value is in service sectors, specifically in hospitality at £4 per m^3. Scotch whisky and livestock farming are two case studies investigated in terms of water use and valuation. Both are key industries for the Scottish economy for different reasons. Scotch whisky is the most valuable product of Scotland after oil and gas and livestock farming produce essential raw materials for the food and beverage industries. The analysis reveals that in Scotch whisky distilleries, the value of water use is much higher than the average value found for manufacturing industries. This can be explained by the high reliance of the whisky industry on the quality and quantity of local water resources. The valuation estimates are transferable to similar locations and sectors after necessary adjustments. The second case study estimated economically efficient yet affordable charge for water use in the livestock industry, and found this to be higher than current charges that apply to both dairy and beef farms.

While valuation is necessary, it is not a sufficient condition to realise full social value. This is where tradability comes in. By making water allocations tradable as far as practically possible, there is potential to reallocate water from low value uses to high value ones. This not only concerns abstractions or volumetric use but also degradation of water quality through different uses, which is an issue more relevant to the current Scottish context. Widespread diffuse pollution caused by agricultural land uses limits the economic availability of water by increasing the cost of treatment for all its users. The thesis considers the options for tradability in a Scottish context with reference to global experience and explores the methods used to allocate and trade water quality rights in relation to potential payment for ecosystem services schemes.

The preliminary results from the optimisation model indicates that while staying within the nitrogen budget set for the case study catchment, farms can maximise their agricultural profit, cut down their fertiliser costs and earn additional income from the sale of their unused allowances for nitrogen pollution at around £1.40 per kg of nitrogen.

Abstract

Valuation draws heavily on the economic theory of demand. This tells us that users have preferences for water and are willing to pay different amounts for units of water put to different uses. Water should be allocated between these uses to the point that equalises the value of the last or 'marginal' unit. In other words, it is impossible to find a higher value for this marginal unit. Application of this principle of equi-marginal returns requires us to have some clarity about water values in competing uses. This is also important since water is rarely free to supply, and therefore suppliers need to charge a price that is in some sense equal to the supply cost and value to achieve full cost recovery.

Even though inclusion of this economic rationale in the management of water resources has been a widely accepted principle, and is included in national and the EU policies, the actual practice does not fully reflect this endorsement. While many countries recognise the vital nature of water resources, few, if any, pursue a rigorous analysis of revealing the explicit value of water as a basis for determining whether water is actually being allocated to sectors in order to maximise its overall benefit to society. Aspiring to be the first Hydro Nation, maximising the social return from its water uses ought to be a policy objective in Scotland.

This thesis constructs a portfolio of different water uses, estimating the approximate value for each and their current allocation in Scotland. This aims to stimulate an informed debate on actual allocation of water among different uses, relative values and trade-offs of these allocations in Scotland so that alternative allocation scenarios can also be discussed. I then focus on the valuation of water by manufacturing industries, the biggest consumptive use and a significant added value creator in Scotland. I investigate the factors that affect the valuation of water and the responsiveness to prices in manufacturing industries using a meta-analysis technique. These values are obviously not the same for each manufacturing sector due the nature of their use and value of their final output. Some sectors create premium value out of their use. The whisky industry stands out as a water-intensive and high value creating sector, as well as a vital contributor to the rural and overall Scottish economy. It is analysed here as the first case study using water footprint and marginal productivity analyses methods, both analyses highlighting the importance of quality and quantity of local water resources in Scotland and its value to the industry. The second case study is the livestock industry, which has been overlooked in the valuation of water use literature yet is significant for livelihoods in rural Scotland where reduced land capability limits agricultural production options.

Following the portfolio of water uses, meta-analysis and case studies that analyse the current situation of value and allocation, I explore how the current situation can be improved through the application of tradability. Currently the main problem in Scotland is not the amount of water used or abstracted, but the pollution reaching water bodies as the result of run-off and leaching from agricultural fields. Therefore, the feasibility of trading water rights is more concerned with the permits to pollute rather than the rights to use. Using a linear optimisation I look into the potential of designing a payment for ecosystem services scheme based on tradability of water pollution in agricultural catchments that are affected by from diffuse pollution. The results indicate that trading schemes help reduce the cost of pollution to all users while creating additional income for farms. For constructing more precise pollution rights and robust schemes more research efforts are required.

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List of Abbreviations

AHDB	Agriculture and Horticulture Development Board
BOD	Biological Oxygen Demand
CD	Cobb-Douglas
CS	Cross Sectional
CWU	Crop Water Use
Defra	Department for Environment, Food and Rural Affairs
DPMAG	Diffuse Pollution Management Advisory Group
EC	European Commission
ECGF	European Container Glass Federation
EEC	European Economic Community
EPA	Environmental Protection Agency
EQS	Environmental Quality Standard
EU	European Union
FAO	Food and Agriculture Organization
GBR	General Binding Rules
GDP	Gross Domestic Product
GVA	Gross Value Added
HGCA	Home-Grown Cereals Authority
ICWE	International Conference on Water and the Environment
IEEP	Institute for European Environmental Policy
IOA	Input Output Assesment
LCA	Life Cycle Assessment
LFA	Less Favoured Area

LPA	Litres of Pure Alcohol
MBI	Market-Based Instruments
ML	Million litres
MLPA	Million litres of pure alcohol
MPA	Marginal Productivity Analysis
N	Nitrogen
NIRAMS	Nitrogen Risk Assessment Model for Scotland
NO_3 mg/l	Nitrogen concentration
NO_3-N mg/l	Nitrate concentration
NVZ	Nitrate Vulnerable Zones
NZ $	New Zealand Dollars
OECD	Organisation for Economic Co-operation and Development
OLS	Ordinary least squares
ONR	Office for Nuclear Regulation
PED	Price Elasticity of Demand
PES	Payments for ecosystem services
PHC	Per Household Consumption
QMS	Quality Meat Scotland
RBMP	River Basin Management Plan
RSPB	Royal Society for the Protection of Birds
SAC	Scottish Agricultural College
SCSG	Spey Catchment Steering Group
SEPA	Scottish Environment Protection Agency
SIC	Standard Industrial Classification
SNH	Scottish Natural Heritage

SNIFFER	Scottish and Northern Ireland Forum For Environmental Research
SRDP	Scottish Rural Development Programme
SRUC	Scotland’s Rural College
SSA	Scottish and Southern Energy
SSBS	Scottish Suckler Beef Support
SW	Scottish Water
SWA	Scottish Whisky Association
SWBS	Scottish Water Business Stream
UKGW Forum	UK GroundWater Forum
UKWIR	UK Water Industry Research
UN	United Nations
USDA	United States Department of Agriculture
WF	Water Footprint
WFD	Water Framework Directive
WFN	Water Footprint Network
WHO	World Health Organisation
WIC	Water Industry Commission
WQT	Water Quality Trading

Chapter 1 Introduction

1.1 Introduction

Water allocation might not seem to be a relevant question in a country with an abundance of resources, but even in Scotland there are examples of competition for uses, which may be exacerbated by climate change. With increasing temperatures, water demand is expected to increase, as is the frequency of extreme weather events. The seasonal and geographical distribution of water is also projected to change drastically, forcing scientists and policy makers to find and implement innovative ways to adapt to and mitigate the risk of imbalances between supply and demand.

The uncertainty in future water availability poses a risk not only to society, but also to the Scottish economy. Many strategically important industries, such as tourism, food and drink, manufacturing and renewable energy generation have been developed around the availability of abundant and good quality water resources. Therefore, any mismanagement of water will have not only environmental but also economic consequences in Scotland. Irrespective of climate scenarios, a country should seek to maximise the social returns from its natural assets including water resources.

Water institutions are influenced by the general cultural, social, economic and political contexts (Saleth and Dinar, 2005). The economic significance of water is reflected in wider state policy. As well as ensuring that the water environment is protected, the Scottish Government is committed to developing the value of their water and improving the productivity of the country's water industry. The Hydro Nation agenda specifically addresses the utilisation of Scottish expertise to maximise the economic benefit of abundant water resources within a sound ecological context by improving efficiency (Scottish Government, 2016a).

The introduction of economic efficiency principles in Scotland dates back to 2003 (McKibbin, 2016) when the Water Framework Directive (WFD) (European Commission, 2000) was transposed into national law. The European Union (EU) legislation had a positive impact on water policy in Scotland (SNIFFER, 2007; IEEP, 2013) in terms of cost recovery and competition in the provision of water-related services by imposing tariffs on consumers (European Commission, 2000). The Water Services Act, that came into force in 2005, has allowed competition in "retail services”. Scotland became the first country in the world to offer its 130,000 non-household water users a choice of water supplier in 2008 (WIC, 2017).

Regardless of whether Scotland remains subject to the WFD post Brexit, there is a need for adaptive water management that recognises the economic aspect of water and its use, and maximises the returns to society from use of water resources. The literature that considers the economic value of water recognises two related attributes that need to be revealed in order to optimise the value of water to any society. The first is the total valuation of water uses (Aylward et al., 2010) and the second relates to its tradability (Layton, 2016). Tradability does not only refer to the actual water rights and volumetric water transfers but also in form of virtual water trade (Allan, 1998), water used to produce goods and services. This is measured with water footprint concept (Van der Zaag and Savenije, 2006).

This thesis explores how this might be done in Scotland by exploring the current allocation of water among the various uses, how much value is created in each, and the benefits of moving towards an allocation that maximises full social value under increased competition for water. It does this with reference to measures and policies on water management and climate change adaptation, and with reference to the ways that valuation can inform and incentivise agendas, such as payment for ecosystem services schemes.

The primary research aim is to understand the trade-off between competing water uses and to challenge the current allocation of water rights that are not necessarily distributed by economic efficiency. This will be done by providing an up-to-date overview of different water demands and their values in Scotland. The secondary aim is to propose a trading mechanism to transfer water use from the low value users to high value users in order to increase the overall return from water uses at a catchment scale.

Section 1.1 explains the introduces the research carried out in the thesis and Section 1.2 provides the motivation for the research aims, Section 1.3 positions the work in the relevant literature, Section 1.4 provides an overview of the structure and content of the thesis in relevance to the research objectives and Section 1.5 discusses the transferability of results beyond Scotland.

1.2 The Motivation for the Research Aims

Water is a scare resource and this scarcity can manifest itself in different forms (Van der Zaag and Savenije, 2006). Physical scarcity is an issue in respect to spatial and temporal variation in water availability, falling short of water demand. Economic scarcity relates to the infrastructure or treatment costs that can make use of available water in a region disproportionately expensive (Brown and Matlock, 2011). Access to clean and sufficient amounts of water is fundamental to the maintenance of human life and ecosystem services (IUCN, 2003). But once these basic requirements are met, in any other instances, water has definite uses of an economic good in which its efficiency of use and social benefit can be maximised through competitive allocation (Atapattu, 2002).

Increasing demand for conflicting and complimentary commercial water uses brings forth the concept of opportunity cost in water allocations (Jaeger et al., 2013), that needs to be managed adaptatively to climate change. Revealing opportunity cost of a certain water use requires the incorporation of economic considerations into water management, a notion widely held and partly implemented in government policies since the Dublin Conference on Water and the Environment (ICWE, 1992) and the United Nations Conference on Environment and Development in Rio (UNCED, 1992). The efficient and sustainable delivery of water services to society requires the implementation of economic tools in the decision making.

As a novel addition to previous water policy in the EU, the Water Framework Directive (WFD) introduced the use of economics in its design and obliged the Member States to implement these principles and to deliver outputs within a strict timeline (Gómez-Limón and Martin-Ortega, 2013; Boeuf and Fritsch, 2016). The WFD required the EU Member Countries to adopt River Basin Management Plans (RBMP) and Programmes of Measures for each RBD starting from 2009 and to update these plans every six years. RBMPs were to include an analysis of river basin characteristics, a review of the impact of human activity on the status of surface waters and on groundwater, and an economic analysis of water use (European Commission, 2000). To this end, various economic considerations for water governance were mandated in several articles of the WFD: disproportionality of costs and cost–benefit analyses (Article 4), economic analysis of water uses (Article 5), cost recovery (Article 9), and cost-effectiveness analysis (Article 11) (Berbel and Expósito, 2018).

In the years following the WFD, the European Commission has made major efforts to operationalise this economic dimensions of The WFD in its actual imlementation by supporting and providing non-binding guidelines on the development of economic instruments (WATECO, 2000). However, these efforts have not yielded the targeted outcomes according to the WFD Implementation Reports, and the Commission's 'Blueprint to safeguard Europe's Water Resources' (European Commission, 2007, 2009, 2012a, 2012b).

The reasons for the partial failure to operationalise economic principles are due to the lack of adequate knowledge capital regarding water economics, barriers to acceptance and lack of consensus about methodologies and concepts among member states (European Commission, 2012a; Maia, 2017; Rey et al., 2018). Implementation processes for economic analysis and instruments have not been straightforward in many member states. Terms such as "disproportionate costs/expenses" and how they relate to affordability were ambiguous. This allowed the states that did not want to contribute fully to justify their exemptions during negotiations (Boeuf et al., 2016).

Issues directly or indirectly related to water policy, such as the impact of agriculture on water quality and quantity, supply demand imbalances and water allocation mechanisms, would have benefited from further and more integrated economic analysis (European Commission, 2012a; Berbel and Expósito, 2018). Excessive abstraction impacts 10% of surface water bodies and 20% of groundwater bodies throughout the EU, and pollution from agriculture increases the cost of water treatment significantly. This limits options for different economic activities in certain regions (European Commission, 2015). As an overall outcome, the WFD target of achieving good ecological status by 2015 was met only in 53 % of surface water bodies in Europe (EPA, 2017; Voulvoulis et al., 2017).

Apart from policy, physical realities also mandate the need to apply an economic approach to water. Climate change will have consequences for all countries, even those without any water shortages. Scotland is expected to experience pronounced changes in seasonal and local water availability. According to observations recorded between 1914 and 2004, Scotland has become wetter and warmer since 1961. Heavy rainfall events have increased significantly in the north and west during winter, with a 60% increase in average rainfall (Scottish Government, 2008). UK Climate Projections from 2009 indicate that summers in Scotland will be warmer and drier while winters are expected to be snowless and wetter.
The region will be affected in different ways and will face diverse kinds of problems. While there may be as much as a 40% reduction in rainfall in the south and east, a continued increase in rainfall is expected in the north and west. The overall climate will

become increasingly unpredictable with more frequent and extreme weather events, such as heavy rainfall, drought and high winds (SNH, 2009; Brown et al., 2012).

These climatic changes and uncertainties cause concerns about seasonal and regional availability of water in Scotland where the demand and supply do not overlap. Water demand increases during the warmer summer months in the Central Belt and East coast where population and economic activities are concentrated, whereas higher precipitation and water availability is experienced in Northern Scotland and the West coast. As bulk transfer of water is costly and energy intensive, water is ideally managed at catchment level. The most efficient way to maintain and increase returns from water resources under reduced availability is to distribute it based on economic efficiency, shifting from low value uses to high value uses.

In this regard, valuation of different uses is necessary as the initial step, but is not a sufficient condition to realise full value- this is where tradability comes in. By making water allocations tradable (as far as practically possible), it is potentially possible to reallocate water use rights between high and low value uses. While suggesting trading, many of the inherent constraints in water reallocation must be recognised, not least the need to guarantee fair shares and pre-existing user rights over some resources.

In Scotland, there are 4,600 km of rivers, 300 km^2 of lochs, 80 km^2 of coastal waters. and more than 100 river basin catchments as well as groundwater aquifers, which feed rivers, and in turn are adversely affected by diffuse pollution mainly arising from rural land use agricultural activities, such as crop and livestock production (DPMAG, 2011). The main pollutants that cause diffuse pollution are nutrients (nitrate and phosphate), suspended solids, faecal bacteria and pesticides. Of these pollutants, nitrate is the greatest threat to human and ecosystem wellbeing and has the longest retention time in groundwater aquifers, causing great social cost (WHO, 2000; Keeler et al., 2016).

Although Scotland has a diffuse pollution strategy (Figure 1.1), it can be criticised for neglecting the synergies and feedback loops between environment, policy and economic activities that cause pollution at a catchment level. For instance, the diffuse pollution strategy currently lacks inclusion of economic instruments beyond economic support and incentives, such as cross-compliance, funding from the Scottish Rural Development Programme (SRDP), and the restoration fund. However, paying farmers through such incentive schemes to stop pollution means rewarding the polluters and is unfair to farmers that are already not polluting (Shortle, 2012). In addition, such direct payments do not reduce the cost of pollution control for the regulators due to the monitoring requirement to identify the polluting farmers, and whether those in receipt of subsidies have reduced their emissions.

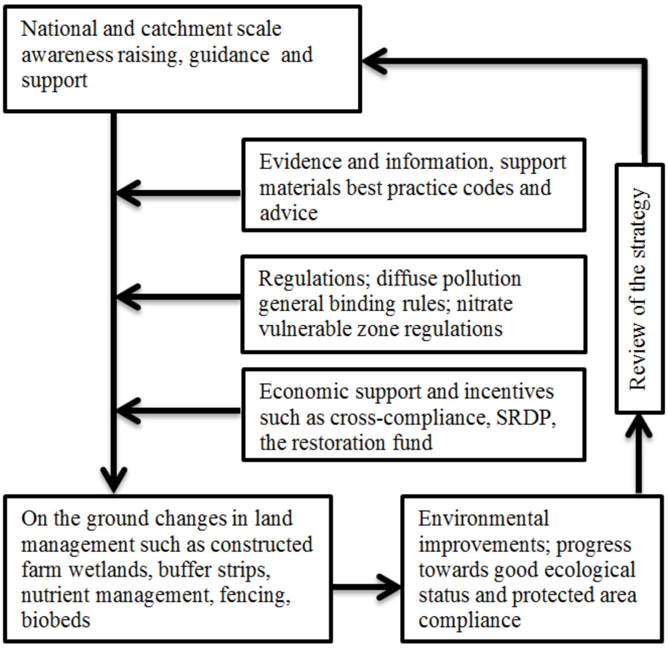

Figure 1.1. Diffuse pollution strategy and its target implementation in Scotland (DPMAG, 2011).

The inclusion of market-based instruments (MBI) in the strategy could help achieve better results in pollution control and a reduction in monitoring costs for the regulators. MBI are applied to the management of water (and other environmental resources) with the aim of complementing traditional policy options in accomplishing a certain policy goal, such as increasing environmental quality or promoting an efficient allocation of water among its users.

1.3 Positioning in the relevant literature

Water resource allocation has been considered by environmental and ecological economists. Environmental economics and ecological economics share the common overarching objective of understanding human-environment interactions and to deliver resource use efficiency. However, in pursuing this common objective, these two sub-disciplines differ in their conceptualisation of environmental resources, utilise different analytical frameworks and have fundamental theoretical and methodological differences. These differences are mainly in the treatment of nature and concepts of resource scarcity and maintaining capital stock, role of technology, population and consumption, equity and welfare (Venkatachalam, 2007).

Environmental economics derived from welfare economics following the work of Pigou (1920) that used theconceptualisation of extenralities as a problem of resource allocation . Negative externalities were presented as a form of 'market failure' within this neoclassical welfare framework. The following environmental economic tradition was also based on neoclassical economic principle, relating supply and demand to individual rationality and the ability to maximise utility or profit. The other significant advances that environmental economics has contributed to the welfare economics discourse can be listed chronologically as Coasian negotiation solution in resource allocation (Coase, 1960); 'second-best solutions' in the area of pollution control (Baumol and Oates, 1988); non-market valuation within micro cost-benefit analysis (Smith, 1993); sustainable development (Pearce and Turner, 1990) and environmental accounting (Ahmad et al., 1989; Venkatachalam, 2007).

First and foremost critique made by ecological economics is the convexity of preferences assumed by the neoclassical economics. When it comes to nature, people might have non-convex preferences, meaning that some people will not make the choices predicted by theory. For example, they will not trade-off money for environmental goods. The possibility of lexicographic preferences introduces problems for the representation of smooth trade offs suggested by neoclassical demand theory.

In essence environmental economics assumes that economic growth can operate entirely independent from the limits of the physical world and views the environment as a subset of the economy (Daly et al., 2003). Natural resources such as water are valued based on their scarcity as an input to production and 'the environment' acts as a sink that absorbs the resulting waste from production (Daly and Farley, 2004). Degradation or failure ofnatural capital can be compensated by increasing other man-made inputs to production such as technology. This is described as the 'strong complementarity' principle (Seidler, 2002; Palmquist, 2005). Moreover, environmental economics focuses primarily on the efficient allocation of resources rather than their equitable distribution and the scale of the economy relative to the ecosystems upon which it is reliant (Dietz and Atkinson, 2010; Beder, 2011). It assumes the rational response of individuals and organisations to the market mechanisms and thus a perfect allocation can be achieved through markets.

Ecological economics appeared in the late 1980s as a critique of the neoclassical discourse of environmental economics. Its roots date back to Nicholas Georgescu-Roegen's The Entropy Law and Economic Processes published in 1971 (Christensen, 1989; Costanza, 1989; Røpke, 2005; Inkpen and Desroches, 2016).

Ecological economics in essence sees economic systems as embedded in, and supported by, natural systems, which are not simply a factor in, but the foundation of, economic activity (Costanza et al., 1998). Therefore, it assumes 'weak complementarity' between nature and man-made inputs to production (Palmquist, 2005). Thus, economic and population growth, as well as consumption in an ecological economics context are limited by planetary boundaries. By integrating models from ecology, economics and other disciplines when relevant, ecological economics seeks to provide interdisciplinary scientific evidence for preserving the natural world (Capra and Jakobsen, 2017). This interdisciplinarity increases understanding of the real world. However, it cannot overcome political and social barriers.

In contrast welfare-based narrative of environmental economics resonates with the policy making and has a significant influence on the regulatory decision making processes (Beder, 2011). So far tools of environmental economic played an impoartant part in the formulation of environmental policy and helped achieve environmental gains worldwide in topics ranging from wetland conservation to preventation of acid rains (Hahn, 2000). Over years ecological economics has also gradually ended up settling for these purely economic methods borrowed from environmental economics such as cost-benefit analysis and valuation of nature (Costanza et al., 1998), natural assets and services, blurring the lines of what distinguished two disciplines initially (Sagoff, 2012).

This thesis sits in the intersection of environmental and ecological economics and aspires to take in the best of both disciplines. It follows the ecological economics perspective on limits to growth and considers the partial, though not a fully weak, complementarity between water and technology, meaning improvements in technology would increase use efficiency to a degree but can not fully resolve the physical water scarcity issue (Jury and Vaux, 2005). To this end, it assumes an economically rational allocation of water as a resource would increase societal welfare in Scotland, or anywhere else. With this objective monetary valuation techniques borrowed from environmental economics are used in Chapter 2 (value transfer), Chapter 3 (meta-analysis), Chapter 4 (marginal productivity) and Chapter 5 (net back analysis) rather than non-monetary valuation. This choice is due to the policy focus of this thesis and its ambition to produce coherent and accessible results that would speak not only to research also but to policy-making circles. At the same time, the thesis endorses the essential role of ecosystem services in supporting human life and economic activities and thus incorporates elements of ecological economics in the design of the bio-economic model of pollution trading in Chapter 6.

1.4 The structure and content of the thesis

An economically optimal distribution of water assumes equal marginal value across users. However, water is typically not distributed according to its economic value. To maximise the value of water resources as suggested in the Hydro Nation Policy, the current allocation of water and the value of each water use created from this valuation must be known. This is essential in order to realise where the potential for transferring water use lies when competition occurs. At present, it is unclear whether Scotland has the information to consider such water trading and this thesis clarifies the position on use and value. The first part of the thesis reviews who uses water in Scotland and what their approximate valuation of this use is, focusing on key industries where there is a literature gap.

Chapter 2 constructs a portfolio of different water uses in Scotland and their corresponding values. The chapter aims to provide a basis for informed debate on actual allocation of water among different uses, relative values and trade-offs of these allocations in Scotland so that alternative allocation scenarios can be also be discussed.

Most manufacturing industries create high value from its water use and use a considerable amount of the available water especially in developed countries. Although the proportion of water demand from industry in developed countries is declining due to adoption of water efficiency measures and emigration of manufacturing industries from developed countries to developing countries, it is still significant. The valuation of water used by manufacturing industries has also received limited research interest and there are few publically available analyses on this topic. Chapter 3 addresses this gap by synthesising available research in a meta-analysis and investigates the significance of factors that are expected to affect valuation of water use by manufacturing industries and the responsiveness of this demand to changes in pricing. This analysis provides critical information that is useful for pricing policies of non-household water in Scotland.

The valuation of water is not the same for each manufacturing sector. Some create premium value out of their water use. In Scotland, the whisky industry stands out as a water-intensive and high value creating sector, producing Scotland's second highest export revenue after oil and gas (Blackett, 2012; SWA, 2015b) out of its use of local water resources. Most distilleries are located in remote rural areas, meaning that the whisky industry also has a unique role in sustaining rural communities. Whisky production is strictly regulated by UK and EU legislation and is dependent on local freshwater resources as Scotch whisky can only be produced in Scotland. Despite the important role of the industry in the national economy and water being essential to whisky production, research on the use and value of water in the industry is limited. Chapter 4 estimates the water footprint of the Scotch whisky supply chain and marginal productivity of water use to quantify the value-added to the Scottish economy through allocation of local water resources to this industry.

The livestock industry is another critical source of income for rural communities in Scotland as a result of the poor land capability that limits arable agriculture. Although quantification of water use in the livestock sector and its effects on water resources have already received substantial research attention due to the expected increase in global demand for meat and dairy products (Forde, 2016; Worldwatch Institute, 2016), there remains a research gap related to the valuation of water use in the livestock industry. Currently no estimate is available for the water valuation of the livestock industry. Chapter 5 therefore investigates the value of water use on dairy and beef livestock farms in Scotland to provide an estimate for the valuation of current water use in comparison to cost estimates of the different supply options.

While it is useful to create country level estimates, water is a bulky commodity best managed at the catchment level. Among all the regulated activities related to water management, diffuse pollution is currently the principal pressure on Scottish freshwater resources and catchments. Unlike point pollution, which is discharged at a definite point or end of a pipe, diffuse pollution happens across extensive land areas through the leaching of pollutants into surface and ground waters with rainfall, soil infiltration and surface run-off, which makes it harder to control, regulate and recover. As well as being the major obstacle to achieving the good ecological status set by the Water Framework Directive (European Commission, 2000), it is also an economic problem as it increases the cost of treatment, and thus the cost of water for all its uses.

The final part of this research considers the options for tradability in a Scottish context with reference to global experience. Although abstraction rights also need to be reformed in order to achieve water allocations that are consistent with scenarios of climate change, the current problem in Scotland is not the amount of water used or abstracted but the water pollution created. Therefore, the current feasibility of trading water rights is more concerned with the rights to pollute rather the rights to use. As uncertainties and time lags are a complicating feature of pollution transport in soil and water in a diffuse pollution setting, trading schemes between non-point polluters (farms) are rare and harder to design (Kerr et al., 2015).

Chapter 6 explores the theory around allocation of pollution rights by discussing relevant literature, and how to re-allocate allowances to pollute water using a trading scheme that considers uncertainties in the transport of pollution load from farms to boreholes, where samples for water quality measurements are taken. In this chapter a potential payment for ecosystem service scheme is proposed for water quality trading for rural catchments that have diffuse pollution problems.

Chapter 7 highlights the significant findings in the earlier chapters and discusses their implications. The limitations of the research are summarised and further research is identified to address these limitations and to extend the current findings.

The academic contribution of the thesis is to synthesise the current state of water valuation literature and policy and to suggest novel applications of existing principles and methodologies. The thesis reemphasises the theoretical principle of economic efficiency of water use as applied to pricing and tradability of rights in practice at a national scale. Although the exercise is carried out in Scotland, the methodologies and outcomes are transferable to other parts of the world.

1.5 The transferability of results beyond Scotland

Issues related to water management are global and maximising overall value from water resources– e.g. in the context of climate change, must be a priorty for any country. Thus, while the results expected to be achieved in the context of the Hydro Nation (HN) agenda, they are relevant to any part of the world where there is potential to increase social returns from water resources. This transferability of results can be grouped into four major categories: creating a portfolio of water uses; researching industrial water use and its valuation; water use in livestock; and the application of trading to water markets.

The EU aims to implement an economic rationale to water management, and California and Australia provide examples of including economic instruments aimed at addressing severe drought and persistent water scarcity problems. However, implementing economic tools requires consensus and methods that can be accessible to practitioners. The water use and valuation exercise carried out for Scotland in Chapter 2 provides a rough template of how to reveal the current allocation of water use in a country. A basic framework reveals who currently has access to water, how much water is used in each category and its value. This therefore helps to illustrate the opportunity cost of current allocations. Such an accounting practice is applicable to any country.

Although the analysis is carried out using data from a developed country, economically efficient water use discourse can be extended to developing country context where the poor currently pays disproportionately high prices for water. The equity concerns related to charging for household water use can be addressed with the introduction of social tariffs, extempting or subsidising the basic use of low income households. Social tariffs have been in place in UK (Defra, 2012) and other developed and developing countries (Acevedo-Antimil et al., 2011; Gonçalves et al., 2014; Mysiak et al., 2015; Schaefer and Warm, 2015; Szabó, 2015).

Chaptesr 3 and 4 consider the under-reported topic of water use and valuation in industry. The meta-regression of value estimates and price elasticity of water demand in Chapter 3 offers an insight into how responsive industrial demand is to increases in water price, and how this effectsvaluation. Using a cross country sample differentiated in terms of development and climate enablesthe meta-analysis to offer globally transferable results and conclusions.

Chapter 4 considers water demand and valuation dynamics of the beverage industry.Scotch whisky is a key industry for Scotland (SWA and 4-Consulting, 2015), but global demand for distilled spirits, such as gin, vodka and whisky, are also expected to increase in future (Distilled Spirits Council, 2018). Therefore, the analysis in Chapter 4 would be useful in the context of any country that is a mass exporter of distilled drinks and interested in revealing the amount and the added value of water use in their drinks industry. The water footprint of Scotch whisky would also be representative for locations such as Ireland where the local and rain-fed barley or other grains are used as raw material in the production of distilled alcoholic bevarages (Dillion Bass Limited, 2011). Marginal productivity analysis is also transferable to locations of a similar regulatory and technological setting, especially within the EU WFD.

The final two chapters (Chapters 5 and 6) deal with agricultural water uses: volumetric water use in livestock (Chapter 5) and the cost of ambient water quality degradation due to agriculture (Chapter 6). Discussing the re-allocation of water rights based on an economic rationale makes it necessary to estimate agricultural users' ability to pay for their water use so that prices may be adjusted accordingly. Netback analysis is used in the first study to provide a water valuation in livestock agriculture, which could be extended to other locations, especially those with similar development, economic activity and climatic circumstances such as New Zealand, where increased water pollution has become a major problem as a result of intensification of agricultural land uses over the recent decades (Macleod and Moller, 2006; Julian et al., 2017).

New Zealand is also an interesting case for Scotland to consider in terms of removal of agro-environmental subsidies. In. New Zealand these were removed in 1985 (Gouin, 2006), wth a gradual switch to payments for ecosystem services (PES) and other trading mechanisms to support farms in exchange for contributing to environmental policy goals (Greenhalgh and Hart, 2015). While, the water quality trading exercise carried out in Chapter 6 aims to mimick this in the Scottish context, the mechanism designed can be transferred anywhere where water pollution from agriculture is significant enough to make it economically feasible to set up a market to achieve cost efficient pollution control. The expected contribution of the conceptual framework would be higher in locations where the major water source is groundwater, which is the case for half of the world's population. (UNEP, 2012).

Chapter 2 Valuing alternative water uses in Scotland

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The candidate collected the data, designed and performed analyses for allocation and valuation of different water uses, wrote the paper. Co-authors provided support and guidance on the scope and design of the project and contributed to the editing of the manuscript.

Chapter 2 Valuing alternative water uses in Scotland

Abstract: Scotland has an abundance of water in comparison to other (EU) countries, yet there are evident local scarcities and competition among different demands for water in some localities. This suggests an obvious need to adopt a rational allocation of water based on the social value in relative uses. Under the Hydro Nation initiative, Scotland has stated its ambition to be a model of water management, which suggests clarity on the different forms of use and their relative market and non-market values. This paper updates previous quantification and valuation information highlighting where competing uses could be exacerbated by emerging scarcities due to changing economic demographic and climatic factors. We also highlight key data gaps that prevent negotiation between competing water rights and the transfer between low and high value uses.

2.1 Introduction

Scotland is a relatively water rich country in terms of annual water availability and precipitation compared to other European countries. Yet conspicuous local scarcities and competing uses in some localities suggest that in Scotland a rational approach to allocate water according to its social value, the sum of aggregate valuations from different financial and non-financial uses is required to maximise social welfare. In its ambition to become a Hydro Nation (Scottish Government, 2016a), the Scottish Government has emphasised the key role of water resources in economic development and is seeking to improve evidence on the ecological, socioeconomic and cultural significance of water uses. The Hydro Nation initiative is politically prominent and emphasises Scotland's water expertise as relevant to the global grand challenges such as food security and climate change, as well as a variety of explicit global values attached to water stewardship such as human development. More immediate domestic priorities relate to the costs of water supply, changing industrial and land use priorities, and the desire to increase resilience to flooding. All point to the need to develop a more transparent understanding of the users of water and the relative values of alternative uses. To date, the use of explicit water valuation in policy decisions is largely limited to investment and operational costs of water supply. The notional opportunity costs associated with prevailing allocations have been neglected.

Although the concept of value is contested (Martin-Ortega et al., 2013), the Dublin Statement on Water and Sustainable Development (ICWE, 1992) and EU Water Framework Directive (WFD) (European Commission, 2000) suggest that economics, and hence monetary valuation, should be a criterion for the valuation and management of water. A comprehensive assessment of water value requires the reconciliation of the different types of use and the consideration of methods to assign explicit or implicit value to each use category. This information is not systematically collected in Scotland in a form that is readily available to inform policy. This chapter addresses the question of how much water is used and by who in Scotland, updating a previous attempt to account for allocation between different uses (Moran et al., 2007) and the valuation of water use by each use.

Section 2 considers the nature of alternative water uses and summarises the data sources used to derive volumetric estimates of use. Section 3 briefly describes the nature of value associated with each of the broad categories of water use, while Section 4 outlines the valuation approaches applied for estimating comparable values for each use category. Section 5 provides a discussion of the significance of these results, including existing data gaps. Section 6 concludes with observations on what the uses and values imply.

2.2 A taxonomy of water uses

Water has a variety of uses, some of which are mediated by quality, others by quantity and several uses that are mutually exclusive (Figure 2.1). The geographical and seasonal availability of water are also important for its supply, or more specifically, the value that can be assigned to it and the cost of access. Quantification of water is an obvious pre-cursor to the consideration of its contribution to social welfare; this includes the normative distribution of the volume of water use and the value per m^3 in alternative and competing uses.

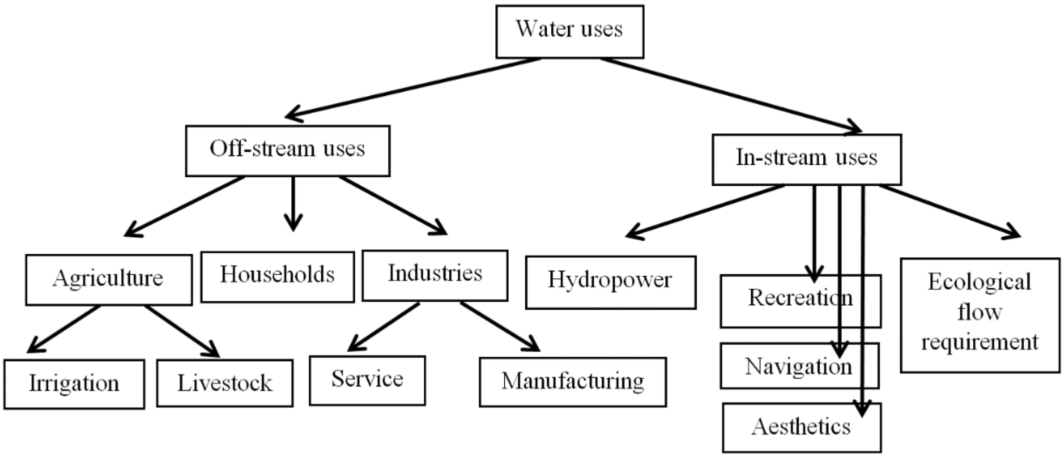

Figure 2.1. Classification of water uses.

Scotland has 16,000 m^3 of exploitable water resources per annum per capita compared to 2090 m^3 in the rest of the UK (Warren, 2002), and received 1757 mm of mean rainfall in 2014, 26.3% above the 1961-1990 average (Kendon et al., 2015; Scottish Government Statistics, 2015). High annual mean precipitation and low population density (67/km^2) contribute to abundant water availability compared to other European countries. The latest water resource forecast indicates considerable headroom between overall supply and demand (Scottish Water, 2014). However, neither population nor water resources are distributed equally across the country; the west being generally wetter, and the east and Central Belt having higher demand due to higher population density, with variation in terms of abundance and quality across different catchments. Climate projections for Scotland predict different levels of water stress across the country, hence the necessary adaptation requirements for irrigation (Committee on Climate Change, 2016).

Access to water in Scotland is regulated by two main bodies. Scottish Water (SW) provides supply and wastewater services to users connected to the mains supply network, and the Scottish Environment Protection Agency (SEPA) issues and regulates abstraction licences for surface and groundwater for private water suppliers. SW is unique both among international suppliers in being publically owned yet benchmarked against private market indicators overseen by an independent economic regulator, the Water Industry Commission for Scotland (WIC). The WIC seeks to introduce elements of market efficiency in supply for retail customers (WIC, 2011). In a sense, this function has already begun the task of introducing a market for water and recognising competing water values.

As long as Scotland remains part of the EU, key legislation governing water use in Scotland includes the EU Water Framework Directive (WFD), which was transposed into Scots law in 2003. The Directive provides a framework for the management and protection of surface and ground waters. More recently, the Water Resources (Scotland) Act 2013 and the Hydro Nation agenda have been implemented to address gaps in overall water management efficiency and to achieve goals beyond the requirements of the WFD. Hydro Nation aims to create an international profile in water management and to add value to the Scottish economy, health, social wellbeing and environment for current and future generations (Scottish Government, 2016a).

Despite these ambitions, specific regulatory challenges remain. A high proportion of use from geographically dispersed private supply sources is a significant complication when accounting for water use and control of water quality. SEPA licensing of withdrawals does not directly translate into actual use, leading to data discrepancies and uncertainty around volumes used. To address this, since 2013, SEPA has had a voluntary updating scheme where licence-holders report their actual annual use so that annual water abstraction amounts can be estimated. However, participation in the scheme is low with only 20% of all registered licences reporting actual water withdrawals.

2.3 Main water uses in Scotland by volume

In the hierarchy of uses (Figure 2.1), the highest level distinction is between in-stream and off-stream uses referring to water either collected from or left in rivers, lakes, groundwater and other reservoirs including both public and private sources. Quality affects some off- and in-stream uses disproportionately (e.g. food and beverage and aquaculture), while quantity is important for others, particularly hydropower generation.

The data used in this paper were obtained from SW and SEPA for 2012-2013 and 2013-2014. The non-household water use information provided by Scottish Water Business Stream (SWBS) and other licensed providers is classified under the Standard Industrial Classification (SIC) codes (UK Government, 2005) in relation to their commercial activity. Each main SIC category was merged with the data from the equivalent SEPA water use classification to enable estimation of total water use by each water user type. In the calculations, the economic level of leakage in the mains water supply is neglected and only volumes used are considered. A linear regression analysis was conducted to fill the data gaps in the reported actual abstraction figures for SEPA licences for uses that might have changing water demand due to weather conditions such as irrigation and livestock. For uses that have more predictable water demands and set processes, such as manufacturing industries and hydropower schemes, the abstraction licences are assumed to be purchased in line with the capacity of facilities and are used fully.

2.3.1 Off-stream uses

Off-stream uses involve the withdrawal of surface or groundwater for a specific use such as public water supply, industrial use, irrigation, livestock requirements or power generation. These off-stream uses are mostly exclusive single purpose in nature and typically of significant opportunity cost, meaning that the use often forecloses other uses. Most extractive uses are consumptive; nevertheless SW assumes an average of 95% of water abstracted is returned to the wastewater collection system or the receiving water body. The remaining 5% can be accepted as the consumed percentage of water and the amount evaporated is not counted. As wastewater collection data are not available for self-supplying establishments (through water abstraction), it is assumed that similar percentages of consumption and wastewater discharge are also valid for the households and industries reliant on private supply.

2.3.1.1 Domestic use

Only 900 households (less than 0.4% of all households) in Scotland have voluntarily switched to metered supply and are charged according to an increasing block tariff for their water use and wastewater collection. For unmetered properties, SW estimates per household consumption (PHC), a measure of average daily volumetric household use, based on a monitored sample of properties within different socioeconomic groups. The average per capita consumption figure for domestic use reported by Scottish Water has been around 150 litres/day in Scotland (Scottish Government, 2017c). Thus, daily domestic water use in Scotland is derived in the range of 799 Mega litres, (Ml, equivalent to 10^6 litres or 10^3 m^3). Annual domestic water use is estimated using this daily figure and the average of country's population between 2012 and 2014 (National Records of Scotland, 2015) is 292 x 10^3 Ml/year.

2.3.1.2 Industrial use

Using the Standard Industrial Classification (SIC), industrial water use can be classified into three major categories of manufacturing, extractive and service industries. Extractive industry is a water-intensive sub-sector categorised as mining, quarrying, milling (crushing, screening, washing, and flotation of mined materials), re-injecting extracted water for secondary oil recovery, and other operations associated with mining activities with higher intensity and a lower productivity compared to other industries. Service industries have water use patterns similar to domestic users, while manufacturing mainly uses water as a direct input or process water in production, in system cooling and for workplace hygiene. We include construction in the manufacturing industries as SEPA private abstraction licences do not distinguish between different types of industries other than extractive ones.

- **Manufacturing industries**

Manufacturing is the second largest sub sector after services in comprising the Scottish Gross Domestic Product (GDP) (SEPA, 2005b). Manufacturing use was estimated as a sum of quantities under the categories of "Industrial or commercial evaporative cooling", "Industrial or commercial non-evaporative cooling", "Industrial or commercial non-evaporative cooling purposes" registered to SEPA for 2013-2014 and entries classified in the SIC in the non-household mains water supply (SWBS, 2014). The estimated volume used in these sectors is 7 x 10^6 Ml/year.

- **Extractive industries**

Mining and quarrying account for 1.1% of employment and 2.13% of the Scottish Gross Value Added (GVA), which is directly linked with GDP and used to estimate the contribution of each individual industry to the economy (National Office for Statistics, 2016). Mining-related water withdrawals tend to be from private supplies rather than mains supply due to the industry requirement for high quantity volume without quality concerns. The total volume used by extractive industries is estimated based on abstraction licences registered to SEPA in 2013-2014 and is 36 x 10^3 Ml/year.

- **Services industries**

Services industry subcategories range from advertising and marketing to hospitality. Establishments that provide personal services such as education, healthcare and hospitality are more sensitive to water availability and quality issues than offices. They have some demand characteristics similar to that of residential water demand, such as a requirement for good quality, while being an average of three times higher in quantity in hotels compared to households (Angulo et al., 2014). Service use was estimated as a sum of actual water use quantities reported for licences registered to SEPA for drinking water purposes under company names for 2013-2014 and entries classified in the SIC category of "All other industrial and commercial" in the "non-household" mains water supply data set of SW for

2013 (SWBS, 2014). The amount of water used by service industries in Scotland is estimated at 78 x 10^3 ML/year.

2.3.1.3 Agricultural irrigation and livestock

Agriculture still plays an important role in the rural economy, although when combined with forestry it constitutes only 1.5% of Scottish GDP (SEPA, 2005). Water demand in agriculture originates from irrigation for crop production and livestock water use.

- **Irrigation**

The irrigation demand in Scotland doubled between 1950 and 2000 (Sylvester-Bradley et al., 2005). Most of this increase has taken place in recent years, mirroring the demand for high-quality produce (Dunn et al., 2004; Murphy et al., 2009) predominantly potatoes but also salad crops and soft fruits, which are grown most commonly in the east to achieve higher yields.

Irrigation holds the largest number of abstraction licences issued. Since producers can obtain abstraction permits at low cost, there is a tendency to acquire more licences than required as insurance against unexpectedly dry summers. Most of the time these permits are not fully used and unexercised water abstractions rights introduce uncertainty to the estimation of water use in the sector.

Irrigation use was estimated by summing non-domestic water supply data categorised for irrigation and SEPA licences registered for “Agricultural irrigation” (Table 2.1). The irrigation licences allocated to golf courses were not considered. The total amount of water used for agricultural irrigation purposes is estimated as 8 x 10^3 Ml/year.

- **Livestock**

Scottish agriculture is heavily dependent on livestock production compared to the rest of the UK and the EU due to reduced land capability. Water in the livestock sector is used for livestock watering, feedlots, drinking, dairy operations, and other on-farm needs. It is assumed that the feed production is all rain-fed and animals do not drink directly from water bodies as they are not allowed to come within 5 m and drink directly from water bodies to avoid pollution (SEPA, 2012). The current water allocation to livestock industry was estimated by summing the abstraction licences registered for “Agricultural other than irrigation" with non-domestic mains water supply allocated to livestock industry related codes (with the exception of aquaculture). The total volume is estimated as 13 x 10^3 Ml/year.

2.3.2 In-stream Uses

This general category usually refers to non-consumptive uses that do not require the withdrawal of the water from its original source, e.g. navigation, hydro-power generation, pollution dilution, freshwater capture fisheries and ecosystem maintenance (Kohli et al., 2010).

2.3.2.1 Environmental flow

The environmental flow is the ecological requirement for water in adequate quantity, quality and seasonality to maintain healthy ecosystems vital to livelihoods (Frankl et al., 2014). Such uses include provision of ecosystem services such as biodiversity, water supply, improved water quality and waste assimilation (the capacity of ecosystems to metabolise a certain amount of pollution without perpetual functional damage) (Leandri, 2009) as well as recreation and aesthetics. Pending consistent flow and quality standards, these uses are largely are non-exclusive.

Maintenance of healthy fish stocks and navigational use can serve as proxy demands for both quality and quantity thresholds (Ecologic Institute and SERI, 2010). An environmental flow requirement of 1,6 Ml/year is calculated by summing water abstraction permits dedicated to environmental services. As no actual use has been reported for water uses registered under this category, we assume all annual allowance through registered permits was abstracted.

2.3.2.2 Aquaculture

Commercial fisheries in Scotland are mostly located on the coast. Therefore, inland fishing in the Scottish context translates to angling and recreational fishing, which is also an economically significant sector (SNH, 2016). However, migratory species such as salmon and trout depend heavily on river habitats and the fisheries licences allocated to abstraction for fishing could be considered to contribute to maintaining ecological qualities fit for fish stock in inland waters. The amount allocated to fisheries is approximately 571 x 10^3 Ml/year, mainly supplied by abstraction and includes also a small contribution from the mains water supply.

2.3.2.3 Navigation

Despite their historical role and considerable potential for economical and low-carbon transport, inland waterways (the Caledonian, Union, Forth and Clyde, Crinan and Monkland canals) in Scotland are used mostly for recreational boating, tourism and property development rather than commercial freight (Scottish Executive, 2011). Given this use, the most appropriate water demand estimate relates to the abstraction licences granted to maintain recreational potential rather than freight or passenger transport. The volume of water allocated to this use is estimated at 82 x 10^3 Ml/year supplied via abstraction licences.

2.3.2.4 Energy generation

Water is used for energy generation in a number of ways, such as cooling water in thermoelectric generation (fossil fuel, nuclear and geothermal) plants and as input in hydropower plants.

Thermoelectricity generation is the most water-intensive and requires large volumes. Use efficiency also varies within the thermoelectric sector between coal powered plants and nuclear facilities and is dependent on the technology used, however on average in a fossil-fuel-fired thermoelectric power plant 95 litres of water is required to produce one kilowatt-hour of electricity (Younos et al., 2009).

Thermoelectricity and hydropower fundamentally differ in their use of water, though both are non-consumptive. Cooling is an extractive use and causes heat pollution in the water used. Hydropower uses the momentum of water (energy output depending on the quantity of water, height of headwater and technology) and is typically,depending on the design,an in-stream use (Frederick et al., 1996). In Scotland thermoelectricity facilities are primarily coastal, which provides access to large volumes of water at a lower cost. Also electricity generation at the Chapelcross nuclear plant, which was located inland, ceased in 2004 (ONR, 2004) significantly reducing the freshwater used in thermoelectricity generation in Scotland. Therefore, the estimate focuses solely on freshwater used in hydropower facilities.

- **Hydropower**

Hydropower generation in Scotland increased significantly by 24% between 2010 and 2014 (Scottish Government, 2016b, 2016c) At present, there are about 120 hydro schemes of various sizes, producing around 43800 TW/year, which is approximately 12% of the overall electricity demand in Scotland. The Scottish Government has plans for more developments across the country as part of its ambition for de-carbonisation and renewable energy implementation. Low costs and reduced CO_2 emissions makes hydropower a favorable option for electricity generation but a considerable amount of water has to be stored and allocated to this non-consumptive use. This potentially competes with other uses including environmental flow, irrigation and fish farms particularly during summer. According to SEPA abstraction licence records, the overall volume of water diverted for hydropower generation annually in Scotland is 150,263 x 10^6 Ml.

The annual water uses by sector estimated as explained above are shown in Table 2.1.

Table 2.1. Water use volumes by use category.

Joint category	SEPA category for private abstraction licences (SEPA, 2015a)	SW SIC category (SWBS, 2014)	Mains supply *	Private abstractions *	Overall annual water withdrawal in 2013-14*
Off-stream uses					
Domestic	Drinking water supply public	Domestic supply	-	-	292,000
Manufactu-ring industries	Industrial or commercial evaporative cooling Industrial or commercial non-evaporative cooling	Food drink tobacco Textile, Clothing and shoes, Pulp and paper Printing and publishing Chemical and petrochemicals Metal and metal products Engineering and machinery Other manufacturing Construction	34,589	6,907,181	6,941,770
Mining and quarrying	Mining and quarrying	Mining and quarrying	19,658	16,651	36,309
Service industries	All other industrial and commercial Drinking water supply private	Wholesale and retail, trade Hotels and catering Financial services Real estate Public sector	72,269	5,704	77,973

Irrigation	Agriculture irrigation fixed point Agriculture irrigation mobile point Agriculture non-classified	Agriculture (Irrigation) Agriculture (Non-irrigation)	2,336	5,653	7,989
Livestock	Agriculture other than irrigation	Livestock	7,214	6,264	13,478
In-stream uses					
Environ-mental flow requirement	Environmenta l service	-	-	1,564	1,564
Freshwater aquaculture	Fish production	-	333	571,399	571,732
Navigation	Navigation including canals	-	-	81,524	81,524
Hydropower	Hydro production	-	-	150,263,442	150,263,442

*all units are reported in Ml/year.

2.4 Valuation methods

Volumetric use data indicates the prevailing distribution of water, which does not necessarily reflect the relative value of alternative uses. These values may be of interest in conditions of scarcity where economic criteria can partly inform an allocation that maximises social welfare- i.e. the sum of aggregate valuations from different financial and non-financial uses. In essence, the aim is to allocate water to its highest economic value, which means that all the value of each water use from different market and non-market perspectives needs to be explicit.

The range of water uses can be valued using different approaches. The valuation of the individual user depends on certain characteristics of a specific water use: the volume and the nature of use. Different market and non-market methods can help reflect opportunity costs of prevailing use patterns.

Economic value of water attached to consumptive and non-consumptive uses can be categorised using a total economic value taxonomy. This attaches direct use value to some uses (i.e. domestic use for cooking), indirect use value (water valued as an input to another market output), option and existence values. The latter move further away from market (or observable financial) information and therefore require non-market valuation methods to help reveal their magnitude.

Non-use values tend to be more contentious and difficult to estimate. A common practice therefore is a value transfer method that uses existing valuation data from previous similar contexts (Koundouri et al., 2013). More robust value transfer information can often be derived from meta-analyses of relevant literature. The meta-analysis enables assessment of variability in values arising from different estimation methods.

The use value of each water demand is estimated using the appropriate technique in relation to the characteristics of the use. Appendix A further explains the suitability of the techniques implemented and key assumptions made for each valuation estimate in the following section.

2.4.1 Valuing off-stream uses

2.4.1.1 Industrial water use

A manufacturing value was derived by a meta-analysis of value estimates for manufacturing and extractive industries to construct a statistically-robust value transfer using purchasing power parity conversion (United Nations, 2016) over the period 1969 - 2014. Final analysis was based on 81 monetary value estimates from eight primary studies, comprising five peer-reviewed articles, one published report and two PhD theses. Chapter 3 of the thesis explains in detail the conduct of the meta-analysis.

The value of water use in service industries was transferred from Angulo et al. (2014) who estimated the shadow price of water use in the Spanish hospitality industry using data collected from 678 establishments over 12 years.

2.4.1.2 Domestic water use

For domestic water use, per m^3 charges for the first 25 m^3 block of water supply supplied and 23.5 m^3 block of wastewater collected by the Scottish Water (Scottish Water, 2013) for metered households in Scotland is summed. The figure is assumed as the representative value of Scottish households' willingness to pay for their water use.

2.4.1.3 Agricultural water use

Irrigation values are transferred from SEPA (2005) which estimated the value of potato irrigation in the West Peffer catchment in Scotland. For livestock use, value transfer was not possible due to a lack of relevant published studies. For this reason, the value for water use on dairy and beef farms was estimated using netback analysis and statistics on commodity markets (SAC Consulting, 2016). Average of the values found in the netback analysis for dairy and beef farms is assumed as the value of water use livestock farmers. Chapter 5 of the thesis explains in detail the conduct of the netback analysis.

2.4.2 Valuing in-stream uses

2.4.2.1 Hydropower

For hydropower, MacLeod et al. (2006) used the cost of electricity production in hydroelectric facilities in Scotland to estimate the long run average value of water allocated to hydropower production compared to other forms of electricity production. The estimates after adjustment to 2015 vary between 0 and 0.0115 £/m^3 depending on whether it is combined with a back-up technology and the technology to which it is compared. This optimal combination option is specific to circumstances of the particular scheme in question. However, the likely option for the general case would be combined cycle gas turbines, which are most compatible with the objective of meeting peak load demands at short notice. Including a level charge for CO_2 set at £10/tonne (MacLeod et al., 2006), this implies a value of 0.0097 £/m^3 to hydropower combined with gas turbines.

2.4.2.2 Navigation

Navigation value was estimated by calculating revenue generated by the issue of boating licences per m^3 actual water abstraction reported to SEPA by Scottish Canals, the authority responsible for managing inland waterways in Scotland (Scottish Canals, 2016). The revenue obtained from boating activities per m^3 of water abstracted by Scottish Canals is used as the proxy of navigational value in Equation 2.1 below:

$$\frac{[(\text{Number } of\ boating\ licences\ issued)\ x\ (annual\ charge\ of\ a\ boating\ licence)}{(Amount\ of\ water\ allocated\ via\ abstraction\ licences\ to\ maintain\ water\ level)} \qquad (2.1)$$

These abstractions contribute beyond maintaining water levels appropriate for (recreational) navigation. However, this approach is limited as it cannot account for the additional in-stream benefits arising from having more water in-stream such as contributing to maintenance of environmental flows. On the other hand, the boaters' full willingness to pay for a boating trip include further expenses such as transportation, food and accommodation, about which we have no local data.

2.4.2.3 Aquaculture

For the freshwater aquaculture industry, a value estimate of 0.00164 £/m^3 is transferred from a previous study that estimated the value of water of use in Scottish aquaculture using the avoided cost method (SNIFFER, 2005).

2.4.2.4 Environmental Flow Requirement

Since no data are available in Scotland for the value of environmental flows, a crude estimate was made based on the value of angling in the Spey River catchment. According to a survey conducted in 2003, the total annual expenditure of anglers visiting the Spey catchment adds up to £1,296,946 in 2015 figures (Butler et al., 2009; Bank of England, 2016). The absolute minimum flow required for fish stocks to survive is determined using the hands-off flow secured by the private Acts of Parliament between 1921 and 1942 (SCSG, 2003).This value is 0.68 m^3/s for the Spey and adds up to a yearly figure of $21.5x10^6$ m^3. In a simplistic approach that does not consider temporal and spatial river dynamics, it can be assumed that the angling income in the region is created by provision of this minimum flow whose unit value is 0.06 £/m^3.

2.5 Results and Discussion

Table 2.2 summarises the values and volumes of water allocated to different uses, estimated by the methods described above. The results should be interpreted within the context for each use considering its specific characteristics.

Table 2.2. Volumes and monetary value estimates for water uses in Scotland.

Water use	Valuation method	Adjusted monetary value (£/m^3)	Allocation in volume (Ml/year)	Value created from current allocation (M£/year)
Domestic	Scottish Water figures for metered households	3	292,000	846
Manufacturing industries**	Meta-analysis	3.68	6,978,421	25680.6
Extractive industries	Meta-analysis			
Service industries	Shadow pricing	4	78,037	312.2
Irrigation	Value transfer	0.25	7,989	2
Livestock	Netback analysis (Own calculation)	1.85	13,478	24.9
Hydropower	Value transfer	0.097	150,263,442	14575.6
Navigation (boating)	Own calculation	0.004	81,524	3.3
Environmental flow	Own calculation	0.061	1,564	0.095
Aquaculture	Value transfer	0.0016	571,402	0.9

*All the figures are converted to 2015 values using inflation rate for £ (Bank of England, 2016). **The final figure for manufacturing industry also includes allocation for mining and quarrying industries.

The consumptive water uses ranked from highest to lowest in volume are manufacturing, domestic, service industries and agriculture (irrigation and livestock combined). By volume, manufacturing industries are the major consumptive use in Scotland. Considering the massive amounts of water required for process and cooling water in most manufacturing industries, this estimation is plausible. Allocation to households is second and is in line with the figures stated in the literature based on population projections for Scotland (Moran et al., 2007). Household demand for water is expected to rise in the future with increasing summer temperatures projected in Scotland.

Water demand in agriculture is heavily dependent on the weather conditions. Farmers tend to buy additional licences to make sure that they secure enough water for an exceptionally dry year and usually do not use all the licences they purchased. However, irrigation demand is projected to grow in the near future as a result of warmer and drier summers and a wider portfolio of high value crops becoming available in Scotland as a result of climate change (Brown et al., 2012). It is therefore expected that abstraction licences allocated to irrigation will then be put to full use.

For non-consumptive uses, the highest ranking by volume is hydropower followed by aquaculture, navigation and environmental flow. The immense volume allocated to hydropower is due to the favourable geographical circumstances for hydropower in Scotland, such as high hills and availability of water, that enable the adoption of hydropower as the one of major renewable electricity source in Scotland (Scottish Government, 2017b). Plans for new pumped storage schemes at Coire Glass and Balmacaan, both 2600 to 5200 TW/year, and for increasing the capacity at the existing Cruachan pumped storage scheme (Nelson, 2013; Poindexter, 2016; SSE, 2017) signal additional allocation to hydropower in the near future.

The estimations are partly in line with the previous assessments and future water use projections of Moran et al. (2007) who estimated water use in electricity generation (including thermoelectricity plants), aquaculture, industries, households and agriculture for the year 2004 and projected for the year 2015. The analyses here give a lower estimate than previous estimates of 56475 Ml/year and 43000Ml/year for irrigation respectively for the years 2004 and 2010 by Moran et al. (2007). This disparity between studies may have arisen from different methodologies. The regression analysis conducted to complete data gaps and assess the actual fraction of use for irrigation and livestock water use categories might have led to underestimation of allocation to these uses in the current study.

Among off-stream water uses, service industries have the highest value per unit of water allocated to them. Value in manufacturing industries, which is £3.68/m^3 is the next highest, followed by households for which the average charge in EU, adjusted by inflation and currency conversion (Bank of England, 2016; Rate Infliation, 2017; XE, 2017) is £2.90/m^3 (Kjellsson and Liu, 2012). The proximity of the value estimation here and the EU average for domestic water charge indicate that the unit value of water is priced in line with its market value. Current livestock water use was found to have a much higher value than irrigation and volumes allocated to livestock are also much higher than to irrigation in Scotland where the livestock industry is a vital part of the rural economy.

In-stream uses generally resulted in low values per unit volume, which is linked with the non-consumptive and non-exclusive nature of these uses. Hydropower has the highest use volume among in-stream uses despite its comparatively low unit value. Current allocation to hydropower creates the second highest overall economic return. Environmental flow requirement resulted in a unit value much lower than hydropower in the analysis, however in a more precise study at catchment level this is expected to be the opposite. The low estimate here could be explained by the consideration of a single ecosystem service (biodiversity) in valuation and the broad scope of the analysis which did incorporate the cost of environmental loss/damage into the valuation.

2.6 Conclusion

Although improving the value of water resources is a priority in UK policy (Welsh Government, 2010; Scottish Parliament, 2011; Scottish Government, 2013), water is often allocated for reasons that have little to do with optimising its value. Revealing value allows the assessment of whether the current allocation is optimal and how it could be improved by reallocation. This study contributes to the literature by providing an overview of the current allocation of water among its uses and their valuation in Scotland using recent data. This overview is a requisite for the discussion on whether the social return from water resources could be improved through reallocation and if water markets could be a feasible option in a future in which more pronounced stresses on water resources are expected caused by climate change. The results have three significant implications.

Increased competition among high volume water uses is expected. While irrigation demand and its value are projected to grow in Scotland (Brown et al., 2012), the hydropower sector will also grow in the near future with planned pumped storage schemes and capacity increases at existing plants. Increasing population and higher summer temperatures mean domestic water demand is also expected to rise (Scottish Water, 2010).

Pricing is the single most powerful policy tool that can improve water use efficiency of Scottish households and industries. Price signals have already proven to incentivise reduction of water consumption by households in various European countries (Biswas and Kirchherr, 2012) with similar welfare and water use levels to Scotland. Encouraging households to switch to metered supply and the implementation of increasing block tariffs for metered households are two policy options to reduce inefficient water use and associated operational expenditures in the household market. Increasing water prices and putting forward policies, such as setting industry level benchmarks for water use efficiency, would incentivise industries to implement measures and adopt technologies for reuse and recycling in order to reduce water related costs.

Increasing demands should not translate into further abstractions as that would put additional stress on surface and groundwater resources and relevant ecosystems. In this regard, re-allocation of water (use) rights currently distributed as private abstraction licences is the first step to increasing the social return from limited resources in hotspots of supply-demand imbalance. Ecosystem services could also benefit from a more flexible and adaptive water allocation through trading that could be changed year on year (Weatherhead et al., 2012).

Whilst non-monetary values are recognised as of key importance to Scotland (Water Resources (Scotland) Act, 2013) in the development of the Hydro Nation approach, this study mainly deals with direct uses that are easier to translate into pricing policy. Additional future research on the non-market perspectives of valuation that produces complimentary results in non-monetary valuation of water use would be beneficial. Further research is also needed to downscale this analysis from a national to a catchment level, considering specific river basin conditions and priorities. Benefits will be twofold. Firstly, sensitivity of ecosystems to water scarcity and irreversibility of the possible damage to natural assets could be assessed more specifically to allow a better understanding of the trade-off between environmental water requirement and other competing high volume uses. Secondly, the demand curves for different industrial users, geographical and temporal factors influencing these demands, and risks and uncertainties associated with unavailability of water or pollution related to specific processes could be better understood and predicted.

Chapter 3 Meta-analysis of economic value and price elasticity in water use by manufacturing and extractive industries

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The candidate collected the data, wrote the regression code, designed and performed analyses for allocation and valuation of different water uses and wrote the paper. Co-authors provided support and guidance on the scope and design of the project and contributed to the editing of the manuscript.

Chapter 3 Meta-analysis of economic value and price elasticity in water use by manufacturing and extractive industries

Abstract: Manufacturing and extractive industries are water-intensive sectors. However, the lack of publicly available data makes it hard to estimate the economic value of water in these industrial uses and its responsiveness to pricing policies measured by price elasticity of demand (PED). Moreover, in the limited literature available on the estimates of PED vary significantly. In this study, we estimated an average value and elasticity estimate transferable to Scotland for manufacturing and extractive industries using previous figures from 26 primary studies for elasticity and 8 primary studies for monetary valuation. Industries analysed as a whole have an average economic value of £3.6/m^3 for their water use, which is much higher than the cost of any possible water supply option in the UK. We used meta-analysis techniques to identify factors that have influenced variations among the primary studies used. The significant factors influencing economic valuation and PED estimates are related to the primary study design and location of the case study. The results highlight the gap between cost and value of water to the industries analysed. They also reveal a foregone opportunity to the public in maximisation of the return from the use of water resources which might have further implications for water pricing policy.

3.1 Introduction

All industries are dependent on water supply for their operations to some extent as an input to production processes, for cooling, producing steam and electricity and for domestic purposes such as sanitation (Wang and Lall, 2002). Estimating an up to date economic value for water demand and understanding its price responsiveness and its affecting factors is important for the design of policies that apply economic principles to the industrial sector and promote higher water productivity by users. Despite the significant water demands of manufacturing industries in most developed countries, there is little research about the valuation of industrial water use and factors affecting the price responsiveness of its users worldwide (Onjala, 2002; Wang and Lall, 2002; Kumar, 2006). The current gap in the literature limits the ability to design such policies in Scotland where information on the valuation of water use by manufacturing industries is particularly important. The manufacturing sector is the biggest consumer of water and second biggest contributor to the Scottish economy, contributing up to 25% of GDP together with extractive industries of mining and offshore oil and gas (Scottish Government, 2016d). Despite this, only one study has specifically examined the valuation of water demand by manufacturing industries in the UK (Rees, 1969), and none in Scotland to date.

In this study, we estimate the value of water to its manufacturing users and the elasticity of their water demand in response to changes in price and identify the factors affecting these estimates. The objectives of this study are twofold. The first objective is to estimate an economic value of water demand by manufacturing industries that would be relevant to the UK and its price responsiveness. The second objective is to improve the general understanding of water demand in manufacturing industries by testing the statistical significance of possible factors that may affect its valuation and price elasticity. This would allow explanation of the causes of variance among estimates in the literature.

Due to time and cost constraints of primary data collection, a value transfer approach has been employed. Meta-analysis has been chosen as it provides more objective and statistically robust results compared to other value transfer methods that are based on a single primary study (Young, 2005). Meta-analysis constitutes a set of statistical tools tailored to synthesise research results obtained in previous studies and the significance of factors affecting the results. Originally developed in the life sciences, meta-analysis is an increasingly popular method to quantitatively survey and synthesise literature in other disciplines (Koricheva et al., 2013) such as economics, especially in the environmental valuation literature (Platt and Ekstrand, 2001; Johnston et al., 2005; Johnston and Duke, 2009; Johnston et al., 2006; Van Houtven et al., 2007; Richardson and Loomis, 2009; Ghermandi et al., 2010). (Platt and Ekstrand, 2001; Johnston *et al.*, 2005; Johnson *et al.*, 2006; Van Houtven *et al.*, 2007; Johnston and Duke, 2009; Richardson and Loomis, 2009a; Ghermandi et al., 2010). Meta-analyses have also been conducted in the valuation of water use in agricultural irrigation (Latinopoulos, 2003, Scheierling et al., 2006; Brouwer and Georgiadou 2011) and in income and price elasticity of household water demand (Espey et al., 1997; Dalhuisen et al., 2003; Sebri, 2014). However, no previous meta-analysis has been conducted for the valuation and/or price elasticity of industrial (manufacturing) water use.

3.2 Data collection and descriptive statistics

The literature review to locate the primary studies for the meta-analysis was conducted by entering relevant keywords in DiscoverEd, the online library portal of the University of Edinburgh as well as relevant online repositories, such as Econlit, Networked Digital Library of Theses and Dissertations, OECD iLibrary, JSTOR, ProQuest Dissertations and

Theses Global, Scopus, Social Sciences Citation Index, Springer, Statista.com, Web of Science Core Collection, Wiley, World Bank: Documents and Reports. Once the initial primary studies were located, additional primary studies were identified in the reference lists of primary studies or potential primary studies. A keyword search on Google was conducted to find observations from grey literature for inclusion in order to minimise the influence of peer-review status of the primary studies on the valuation and elasticity observations. As many observations as possible from different yet valid sources were included from the beginning to eliminate the risk of publication bias and published studies reporting more significant results than others. The full lists of studies used for the meta-analysis regressions are contained in Appendix B.

Two separate data sets were constructed and analysed. One set contained observations of price elasticities, while the second contained observations on the value of water use expressed in £/m^3 after necessary metric conversions. Six of the studies identified included estimates of both price elasticity and monetary value (Wang and Lall, 2002; Rojas, 2005; Kumar, 2006; Ku and Yoo, 2011; Nahman and DeLange, 2012; Tobarra-Gonzalez, 2015). The rest only included either valuation in monetary units (8 studies) or price elasticity estimates (22 studies). The monetary value estimates in different national currencies were first converted to the international US dollar rate in the year of the study, or year of the data set if it was indicated, using purchasing power parity rates expressed for each year (OECD, 2015). The value in US dollars was then converted to the UK sterling rate of the year of study and the inflation rate of the UK pound between the study year and 2015 was factored to update all values to the year in which the meta-regression was conducted. Code in R used in the regression could be found in Appendix C. The conversion was carried out in order to have a uniform unit throughout the sample and to have a UK relevant figure. Results are reported in the unit £/m^3 for consistency with the findings of the rest of the thesis. Conversion was not required for observations of price elasticity as the income and pricing influences are already a part of unitless price elasticity estimation derived from Equation 3.1.

$$\mathrm{PED} = \frac{\Delta \mathrm{lnQ}}{\Delta \mathrm{lnP}} \tag{3.1}$$

PED : Price elasticity of demand

ΔlnQ : Change in the volume of water demand

ΔlnP : Change in the price of water

Initially 159 PED observations were collected (Table 3.1). The observations were first reduced to price elasticity estimates by removing: (1) output elasticity estimates (Ku and Yoo, 2011), (2) observations using the same data set in the estimation of several observations (Rojas, 2005), and (3) studies for which full text could not be found (Williams and Suh, 1986). The remaining price elasticity estimates were reduced to 113 by removing erroneous positive observations (Babin et al., 1982; Nahman and Lange, 2012; Onjala, 2002; Reynaud, 2003; Wang and Lall, 2002). In addition unfeasibly high estimates were removed, e.g. -1489.3 for the paper industry (Onjala, 2002). Observations that reported values for service industries or utilities were also removed to ensure that only manufacturing and mining industries, including offshore oil and gas extraction, were included in the meta-analysis. In the analyses, petrochemical and extractive (e.g. mining, quarrying) industries were also included in the data sample because a significant number of primary studies estimated a value and price elasticity for both.

Originally 128 monetary value observations from 8 peer reviewed primary studies, with the exception of one PhD thesis (Rojas, 2005) and one report (Nahman and DeLange, 2012), were collected for the monetary valuation meta-regression (Table 3.2). Multiple value observations originating from the same primary study over the years (Fujii et al., 2012), or for different regions (Rojas, 2005) reporting a value for the same sub-industry, were aggregated, reducing the dominating weight (around 50 observations from each) of these two studies in the sample. This reduced the final sample to 81 monetary observations.

Each primary study used a different system of categorising sub-manufacturing industries. The categorisation of study results used here in both data sets is explained in Appendix D.

Table 3.1. Distribution of primary studies used in the meta-analysis of price elasticity values by sector.

Sector	Number of primary studies reporting a value for this category	Total number of observations
All manufacturing sectors	9	19
Chemical and allied industries	7	22
Electrical and mechanical industries	3	8
Food and beverage industries	9	38
Mining and allied industries	6	22
Paper and allied industries	9	18
Petroleum related industries	4	5
Textile and allied industry	6	17
Unclassified manufacturing industries	6	10
Total	22*	159 reduced to 112

*Not all studies used the same type of sub-sectorial classification or report values in all the sub-categories used to classify the exact sub-categories of manufacturing industries.

Table 3.2. Distribution of primary studies used in the meta-analysis of economic valuation by sector.

Sector	Number of primary studies that report a value for this category	Total number of observations
All manufacturing sectors	6	27
Chemical and allied industries	5	21
Electrical and mechanical industries	3	13
Food and beverage	2	12
Mining and allied industries	4	9
Paper and allied industries	4	5
Petroleum related industries	4	13
Textile and allied industry	5	15
Unclassified manufacturing industries	5	13
Total	8*	128 reduced to 81

*Not all studies used the same type of sub-sectorial classification or report values in all sub-categories used to classify the sub-categories of manufacturing industries.

Two types of issues affecting the robustness of meta-analyses were checked in the data sets prior to the meta-regression analyses (Dalhuisen et al., 2003; Sebri, 2014). The first is the risk of various biases arising intentionally or unintentionally through data collection and management, and the second is the limitation imposed by the available data sets and observations on the statistical robustness of the analysis. The rest of this section explains how each issue was addressed in the analysis.

After the collection of observations and the construction of data sets for meta-analysis, publication and authorship biases were checked. Publication bias refers to the risk that published articles might have deliberate or inadvertent modification to achieve significant results in order to be more publishable (Brouwer and Georgiadou, 2011). The best way to reduce publication bias is to include both peer-reviewed and non-peer-reviewed work such as reports, dissertations, unpublished manuscripts and other grey literature examples in the meta-database (Song et al., 2012). A funnel asymmetry test can also be applied to the data sample to quantitatively test for publication bias (Higgins and Green, 2011). The distribution of observations is expected to be symmetric around the mean if publication bias is minimal or non-existent in the data sample (Sedgwick, 2013). As a rule of thumb, funnel tests can only be reliable when there are at least 10 studies in the meta-analysis, otherwise the power of the tests is too low to distinguish chance from real asymmetry in the sample (Higgins and Green, 2011). For this reason, only the sample for elasticity could be tested with this method.

We moved the symmetry axis to x=-1 as it is an error to find a positive price elasticity for water demand, the funnel plot for the PED sample (Figure 3.1) is not perfectly symmetric, indicating the possibility of publication bias. We cannot conclude definitively that there is a publication bias as we have considered all relevant literature including grey literature and unpublished work through a systematic review of the available sources as explained at the beginning of Section 2. However, it is also possible that many studies with a negative outcome (no significant effect) are not published in the first place. This also causes bias and that we cannot control. In addition, studies taken from non-reviewed sources contain the most outlying estimates of both PED and monetary valuation as expected (Stanley, 2013).

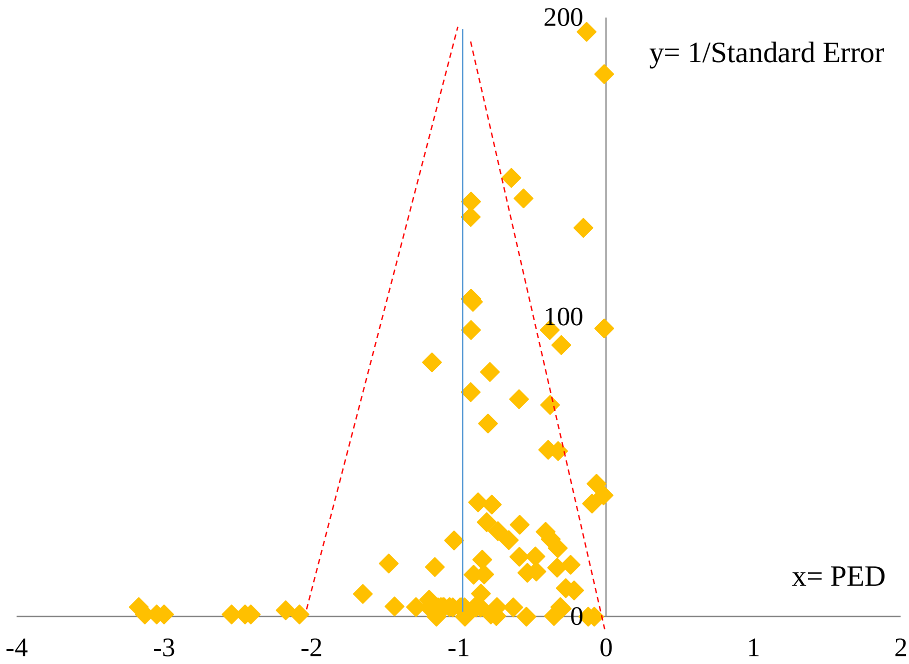

Figure 3.1. Funnel test for PED sample with symmetry axis moved from 0 to -1 and dashed lines showing where the symmetry is expected.

The "study bias", a single study or a small number of studies with a disproportionate number of observations distorting the results, was partially expected in the sample due to the limited number of studies in the monetary valuation data set. To offset this issue, for any study reporting observations from different years for a single manufacturing industry, the mean of the estimates was calculated, a new SD value of the mean of the observations was created and a single observation for the median year in the primary study reported. However, this was not possible for some studies with multiple observations due to their study design. For example, while we were able to aggregate the monetary valuation observations from the same manufacturing industry over a decade in Fujii et al. (2012), it was not possible to aggregate estimates from Ku and Yoo (2011) which used the data set from the same year to analyse variance caused by two different function estimations in the valuation. For this case, it was assumed that half of the sample was tested with one functional form and the other half with the other form to reduce the effect of the same data set being used twice in the estimations of the same sub-industry. Moreover, this bias was tested with an explanatory variable "study code" in the regression.

“Authorship bias”, which is the risk of nesting estimates within sites and within authors as a result of including several studies from the same author (Bateman and Jones, 2003), was also taken into account. In the data sets used here, authorship bias is minimal since only three authors (Renzetti, Reynaud and Rojas) contributed more than one primary study to the data sample. Apart from the observations from Rojas (2005), which are already consolidated, observations from the two other authors accounted for only a small proportion of the observations.

The data set has other limitations. Some studies in the sample did not include any information about the share of sub-manufacturing industry where they reported an elasticity or monetary value estimate for the overall sample used in the study. In these cases, an equal distribution of sample size across different sub-industries had to be assumed, as it was not possible to find external information on the percentage of each sub-manufacturing industry in the specific case study regions for the year of the primary data used.

3.3 Methodology

3.3.1 Valuation and price elasticity estimations

After the necessary currency and inflation conversions and data handling processes described in Section 2, the mean average of the monetary value and price elasticity observations were calculated across the sample. Thus, a final monetary value and price elasticity estimate for each industry were constructed. Initially, to make the context more relevant to Scotland, only observations from developed countries were considered in this average. However, for most sub-industries, there were not enough observations or observations originated from only one study, which meant that the whole sample had to be considered.

3.3.2 Meta-regression methods

Meta-regression analysis can help identify the causes of variance among observations from different studies by statistically testing factors that are expected to have an effect on the willingness to pay for water use by manufacturing industries. The hypothesis is that each factor represented by an explanatory variable important in the valuation of water use, with some being more significant than others in causing the variance among observations. We aim to understand which factors are the main drivers using the results of the regression analysis.

In the meta-regressions, PED and valuation estimates were the dependent variable (y), whilst the following explanatory variables (x) were considered individually (Table 3.3). These 15 factors (explanatory variables) were categorised into four main groups. The first two group of factors were related to the study design (Category 1) and type of data used (Category 2) in the primary studies.

The design of the primary studies were expected to have an influence on the results these studies found (Johnston et al., 2006). Thus, the first two categories of variables target to test if this expected influence also exists in the samples here. These variables define the objective function and data in the primary study are compiled from the former meta-analysis studies that look at the valuation of water use and water quality improvement (Espey, 1998; Platt and Ekstrand, 2001; Dalhuisen et al., 2003; van Houtven, 2008; Sebri, 2013). Due to the size of the monetary valuation sample, only two of the functional types and forms were present and thus the binary variables are used in both samples to make the two regressions comparable. However, in terms of functional estimation a factorial variable is used. Two primary monetary valuation studies (Wang and Lall, 2002; Ku and Yoo, 2011) used two different functional estimation (Cobb-Douglas and Trans-log) to compare the effects of functional estimation. This provided sufficient observations of different functional estimations.

Each study was given a code (listed in Appendix B) and these variables in the form of text were used in the regression as factorial variables to identify if there is any effect of primary study on the elasticity or value observation (Stanley, 2013). Numeric variables (SD, data year, sample size) were used for data in numeric format in the primary studies. The sample size and SD variables were tested to check if the quality of the primary study has an impact on the results as expected. The year of the study was used as a measure to test whether there has been an increase in the valuation of water and the responsiveness of its demand over the years. An increase in the valuation of water use over the last decades is expected to be observed in the data as a result of increased awareness of impacts of climate change and attempts for inclusion of economic principles in the water policy worldwide.

The second two categories of variables were related to the characteristics of the case study (Category 3) and sub-industry (Category 4). The information on explanatory variables “development level” and “water stress” of the case study country in the primary study were not directly reported in the primary studies but are expected to influence manufacturing water demand. Development category assumed that OECD members are developed and the others are developing countries (OECD, 2015). UN World Water Assessment Programme's classification (WWAP, 2012) was used for categorising case study locations by water stress. Together with country and water-stress variables, it was aimed to test the impact of a country's development and climate on the price elasticity and monetary valuation observations in order to test how representative the sample consisting mostly of developing countries would be of the situation in Scotland, even after purchasing power parity adjustments. The "industry type" variable was used to understand whether the nature of water demand in a specific sub-industry affects its valuation (Statistics Canada, 2017). Heavy manufacturing industries and mining and quarrying industries were expected to be more water intensive industries in terms of water use per value added (Eurostat, 2017). Finally, the three level water intensity categorisation (Hough, 2017) was adopted to test the effect of "water intensity" and to see if there is a link between the volumetric water demand of industries and their valuation of water use.

Table 3.3. Explanatory variables used in the meta-regressions.

Variable	Variable type	Regression price elasticity	Regression monetary values
Category 1: Design characteristics of the primary studies			
Function (model) type	Binary	If Marginal productivity analysis (MPA) =1, if Other=0	If MPA=1, if other=0
Functional form	Binary	If Ordinary least ordinary squares (OLS) =1, if Other=0	If OLS=1, if other=0
Function estimation	Factorial	If Trans-log=1, if Cobb-Douglas=2; if Other=3	If Trans-log=1, if Cobb-Douglas=2; if Other=3
Standard deviation	Numeric	-	-
Publication type	Binary	If Peer-reviewed=1, Not reviewed=0	If Peer-reviewed=1, Not reviewed=0
Study code	Text	-	-
Methodology	Joint	-	-
(model+ functional estimation+ functional form+ standard deviation)			
Category 2: Data characteristics of the primary studies			
Data type	Binary	If cross-sectional (CS)* =1, if Other data=0	If CS primary data=1, if other primary data type=0
Date of data set	Numeric	Between the years 1964-2012	Between the years1994-2012
Sample size	Numeric	Between 2 and 2000	Between 3 and 53912
Category 3: Environmental characteristics of the case study location			
Development (OECD, 2015)	Binary	If Organisation for Economic Co-operation and	If OECD country=1, if not=0

		Development (OECD) country=1, if not=0	
Water stress (WWAP, 2012)	Binary	If water-stressed=1, if not= 0	If water-stressed=1, if not= 0
Location	Text	12 countries (most observations from North America)	7 countries (most observations from China)
Category 4: Sub-sector characteristics			
Industry type	Factorial	For both regressions: If all manufacturing industries=1, chemicals and allied=2, electrical and mechanical industries=3, food and beverage industries =4, mining and allied industries=5, paper and allied industries=6, petroleum related industries=7, textile and allied industries=8, unclassified manufacturing industries=9	
Water intensity	Factorial	For both regressions: If intense=1, If medium=2, If neither=3	

* The type of data composed by observing many subjects at the same point of time (here year) are called cross-sectional data.

A variety of statistical methods have been used for meta-analyses, none of which are accepted to be superior by consensus (Johnston et al., 2006). Here the assumption was made that both data sets contained representative samples since they included all available observations identified from extensive literature and online searches.

Initially a mixed-effects regression was considered with weights that handle individual observations with a random effect and the overall sample with fixed effects, assigning weights to each study based on its SD and the number of observations (Riley et al., 2010) (Equation 3.2). This specific methodology was chosen because the weighted approach has been proven to address heterogeneity better than a fixed-effects meta-regression (Benos and Zotou, 2014; Doucouliagos et al., 2012) and to reduce the impact of over-represented studies in the sample.

$$\text{Weight of each study in the sample} = \frac{1}{(n \times SD^2)} \qquad (3.2)$$

where:

n: number of observations in the primary study
SD: Standard deviation value reported in the primary study

However, due to data sets and chi-test results, this option was not adopted. Instead, to capture the individual influence of each explanatory variable, a fixed-effect linear regression model was used. The meta-regressions for both data sets were first checked for the individual effect of each variable and then the group of variables within each category. The generalised function of the regression is represented in Equation 3.3.

$$y_i = \alpha_0 + \sum_{k=1}^{K} \alpha_k X_{ik} + \varepsilon_i \tag{3.3}$$

where:
y_i: dependent variable, observed estimates of elasticity or standardised monetary values
i (1….M) : primary studies
k (1…K) :total descriptive variables
α_0 : the intercept
α_k : meta-regression coefficient of k^{th} descriptive variable
X_{ik} : variables that explain variation in the estimates across the studies
ε_i : regression residuals (under the assumption of normal distribution and variance r^2).

Ordinary least squares (OLS) was used as it has been applied in several previous meta-analysis studies (Loomis and White, 1996; Rosenberger and Loomis, 2000; Zamparini and Reggiani, 2007). The choice of fixed-effects regression limits the choice of the estimation technique to least squares based maximum likelihood approaches (Gelman, 2005). One disadvantage of the OLS method is that it might yield biased estimates towards studies that provide a greater number of observations in the sample, which are non-independent. or where observations are correlated. This might cause issues of heteroscedasticity and variance estimation in the meta-regression error terms (Dalhuisen et al., 2003).

To address this issue, the data set was reformatted as already discussed in Section 2 to reduce the weight of certain studies (Rojas, 2005; Ku and Yoo, 2011; Fujii et al., 2012) and the resulting heteroscedasticity as much as possible. The number of explanatory variables was also reduced to the 14 shown in Table 3.3 by removing those that were not as representative or were likely to be auto-correlated with other explanatory variables, such as water productivity, which is directly linked with type of sub-industry or supply type.

3.4 Results

In this section, we discuss the findings of PED, monetary value estimate and meta-regression.

While PED demonstrates how users of water are likely to respond to changes if the prices are increased and is directly relevant to the influencing conservation behaviour (Hökby and Söderqvist, 2003), the upper boundary of how much they can pay is measured through their willingness to pay which is estimated through economic valuation.

The basics of interpreting PED can be summarised as below:

i) PED= ∞ Perfectly elastic demand.

ii) ∞> PED >1 Relatively elastic demand

iii) PED =1 Unit elastic demand

iv) 0< PED < 1 Relatively inelastic demand

v) PED = 0 Perfectly inelastic demand

PED is negative according to the theory of demand (Espey et al., 1997; Dalhuisen et al., 2003). Therefore, in price elasticity metrics, the absolute value of the number, the magnitude of its distance from zero, is used to interpret PED. The higher the absolute value of PED, the more elastic the demand is, meaning the more sensitive consumers are to price changes. Table 3.4 shows that the PED in manufacturing industries ranges from relatively inelastic (all manufacturing industries, chemicals and allied industries, food and beverage industries) to relatively elastic (electrical and mechanical industries, mining and allied industries, paper and allied industries, petroleum related industries textile and allied industries and unclassified manufacturing industries).

Table 3.4. PED and monetary value estimates.

Industry type	**PED estimates**	**Monetary value estimate (£/m^3)**
All manufacturing industries	-0.8	3.6
Chemicals and allied industries	-0.9	3.0
Electrical and mechanical industries	-1.8	4.1
Food and beverage industries	-0.8	0.8
Mining and allied industries	-1.1	3.6
Paper and allied industries	-1.4	1.7
Petroleum related industries	-1.8	12.0
Textile and allied industries	-1.4	4.6
Unclassified manufacturing industries	-1.3	3.0

Unfortunately, due to insufficient levels of detail on the pricing, tariff and supply characteristics in the primary studies, the distinction between short and long term PED could not be specified. We expect long term price elasticity of water, as of other utilities (Filippini, 2010; Jamil and Ahmad, 2011; Sita et al., 2012; Havranek and Kokes, 2015), to be more important in manufacturing facilities due to fixed contracts, as changes in the water supply price cannot be offset by the immediate reduction of output or change in technology in the short term (MacLeod et al., 2006; Musolesi and Nosvelli, 2011; Hortova and Krištoufek, 2014).

The economic valuation results here are highly biased by the primary study and its case study location due to the very limited number of primary studies. Overall, the monetary value estimate for water for manufacturing industries (3.6 £/m^3) is consistent with the valuation literature (Wang and Lall, 2002; Kumar, 2004; Ku and Yoo, 2011; Eurostat, 2014; Tobarra-Gonzalez, 2015). The highest unit value (12 £/m^3) is for the petroleum industry. Considering the profits of the UK petroleum industry and its mostly offshore locations (Hough, 2017) where desalination, one of the most expensive forms of water supply (Karagiannis and Soldatos, 2008; Adham, 2015), is still economically feasible as the main source of water, the estimate is reasonable. However, other high value water users, such as the food and beverage industry, yielded unexpectedly (Mathieson et al., 2002; Eurostat,

2017) low values compared to textile or paper allied industries. This can be explained by several reasons, such as the limited number of primary studies, the non-transparent classification of industries in the primary studies (Inthout et al., 2015; Ioannidis and Roberts, 2018), and clustering of some locations and publications in certain sub-industries. For instance, of the 10 observations in the food and beverage industries, only 2 are from a developed country and these 2 originate from the same study (Ku and Yoo, 2011).

Given that most explanatory variables are dummy or factorial variables, the meta-regression results (Table 3.5) should be interpreted based on their signs and statistical significance via p-values, rather than their marginal estimation values in the regression (Espey, 1998; Dalhuisen et al., 2003; Ioannidis and Roberts, 2018). The consistency of the signs and statistical significances (indicated in bold in Table 3.5) can be further interpreted for the robustness of the analyses (Espey et al., 1997).

The meta-regression was based on the a priori hypothesis that certain factors affect the valuation and price elasticity of water demand in manufacturing industries and these factors cause variance in estimates. This was partially confirmed, as many explanatory variables listed in Table 3.5 are not significant, and the factors that are significant are not necessarily the same ones for the PED and monetary valuation samples. This inconsistency could simply be a result of insufficient data and inherent issues with the estimation method.

Table 3.5. Results of meta-regression analyses of factors affecting price elasticity and monetary valuation estimates.

Descriptive variable	**PED data**		**Monetary valuation dat**	
	Estimate (SD)	**p-value (t stats)**	**Estimate (SD)**	**p-value (t stats)**
Category 1: Design characteristics of the primary studies				
Function (model) type	0.529 (0.318)	**0.009 (1.664)**	5.962 (2.018)	**0.009 (2.659)**
Functional form	0.170 (0.220)	0.9 (0.015)	2.516 (1.833)	0.174 (1.372)
Function estimation	2.516 (0.094)	0.072 (1.808)	7.812 (1.336)	**<0.001 (5.847)**
Standard deviation	0.362 (0.108)	**0.001 (3.339)**	0.4667 (0.496)	**0.018 (2.434)**
Publication type	0.536 (0.226)	**0.018 (5.611)**	2.706 (1.974)	0.173 (1.371)
Study code	1.3780 (1.0030)	0.1193 (2.438)	27.346 (2.354)	**<0.001 (11.617)**
Methodology (model+ functional estimation+ functional form+ standard deviation)	1.378 (1.003)	0.1153 (1.374)	24.681(0.42)	**0.004 (0.170)**
Category 2: Data characteristics of the primary studies				
Data type	0.455 (0.244)	0.065 (1.858)	5.241 (2.027)	6.688 (0.011)
Data year	0.275 (0.146)	0.064 (1.876)	-0.275 (0.146)	0.645 (18.76)
Sample size	0.0003 (0.0001)	0.24 (1.389)	0.000 (0.000)	0.276 (1.113)
Category 3: Environmental characteristics of the case study location				
Water stress	4.815(1.877)	2.566 (6.854)	0.465 (0.255)	3.329 (1.824)
Development	0.739 (0.265)	0.0698 (2.780)	18.080 (2.363)	**<0.001 (7.651)**
Location	1.732 (0.531)	**<0.001 (1.958)**	15.950 (5.825)	**0.007 (7.651)**
Category 4: Sub-sector characteristics				
Industry type	0.649 (0.292)	0.491(0.953)	1.574 (2.893)	**<0.001(2.2e16)**
Water intensity	0.17 (0.280)	**0.048 (3.607)**	2.153(24.398)	0.678 (0.172)

Among the explanatory variables that were tested here, model type, standard deviation and location are significant for both PED and valuation. Explanatory variables “model type” and “standard deviation” respectively confirm the hypotheses that using the marginal productivity method affects PED and valuation estimates, and that observations with lower standard deviation figures are more consistent with the overall sample. The significance of the location variable reveals that there might also be other factors beyond those accounted for in economic valuation studies, such as legislation, governance, power relations and influence of lobbies (Bryant and George, 2016) on access to water or policy priorities in its allocation (Tsur and Dinar, 1995; Ruijs et al., 2008), that cause variance in the observations.

On the other hand, the hypothesis that explanatory variables related to data used in the primary study (data type, data year or sample size) have an effect on the estimates is refuted as these variables are not significant. This can be explained by the non-transparent handling of data in the primary studies.

Variables of study design, such as “function estimation”, “study code” and “methodology”, were significant for monetary value, but not for the elasticity meta-regression. This can be explained by the more balanced sample across studies in the price elasticity data set which has more primary studies. Therefore, the regression for “study design” variable was less influenced by the each primary study that the observations were sourced from (Stanley, 2013; Inthout et al., 2015). The term “study code” tests for study bias by which assuming that the primary study containing the observation has a considerable effect on the estimate and that observations from the same source tend to cluster together (Bateman and Jones, 2003). The likely presence of study bias (Bateman and Jones, 2003) was confirmed by its significance in the monetary values regression, where the effect of the primary study was expected to be much more pronounced because of the very limited number of primary studies available.

3.5 Conclusions

Appropriate pricing of water is critical for improving its efficient use and sustainable management, both of which require a good understanding of influences on water demand. However, the limited number of published studies on the valuation of water use by manufacturing industries prevents such efforts. Considering the volumes consumed and values created in these industries, the assumption is made that the current state of information on water use and value is a public access issue rather than an availability issue.

Due to competition and other legal concerns, the companies or the industries might not want to share their resource use information and choose to keep the estimations they have conducted confidential. By synthesising the available empirical literature on the manufacturing water demand, this study provides updated figures for price elasticity and valuation of water demand adapted to the UK through the purchasing power parity technique, as well as a meta-analysis of factors that are assumed to cause variation in estimates in both. The low number of available studies reduces the robustness of the analysis here, especially of the monetary valuation.

The Scottish Government is committed to maximising the value of water resources as national assets as a part of the Hydro Nation Agenda (Scottish Government, 2016a).The Hydro Nation policy should be based at least partly on a principle of realising value from a natural resource. At present, it is unclear whether Scotland has the information to consider such a move and this study aims to stimulate an informed debate on water use in different manufacturing and extractive industries and their relative values. In terms of benefit and final impact, this study also contributes to the Hydro Nation policy by providing publicly available figures that could be used as proxies despite certain shortcomings as detailed above.

The results here have several implications. Firstly, a pattern in PED can be identified between different types of industries. While industries that use water only in production processes and cooling (electrical and mechanical industries, mining and allied industries, paper and allied industries, textile and allied industries) have relatively elastic demands, industries that also use water as direct input to production (chemicals and allied industries and food and beverage industries) have relatively inelastic demands. For the first group of industries, there is a higher possibility to reduce water use by means of a substitutability relationship between the choice of technology or other production inputs without compromising output levels. Thus, the effect of pricing on the water conservation behaviour of such industries is expected to be higher.

Secondly, industries analysed as a whole have an average economic value of 3.6 £/m^3 for their water use. This is much higher than the cost of any possible water supply option in the UK. The gap between cost and value of water to the industries analysed reveals a foregone opportunity to the public in maximisation of the return from the use of water resources. To help realise this opportunity, a different shadow price for each sub-manufacturing industry linked with the value they create from this use could be considered in allocation decisions under competition and when estimating their willingness-to-pay for water supply.

Due to the unavailability of information in most of the primary studies, it was not possible to test the effect of water supply type and tariff type for the mains water use on the elasticity of the water demand in the meta-analysis. The industries dependent on mains supply and increasing block tariffs expect to have a higher tendency to opt for investment in technology to reduce water related costs (Reynaud and Thomas, 2013). Further investigation into the type of supply and how it is charged will be a valuable complement to the findings of this study in understanding consumer behaviour and substitution between different types of supply (mains versus private supply) and technology (normal supply versus recirculation or desalinisation).

Chapter 4 Water use in the Scotch whisky industry

The candidate is the main author of Chapter 4. She conducted the literature review, identified the gap and research question for both analyses, and gathered the primary data. She wrote the chapter, including the discussions in the results section. The final data sets and results from the analyses are cited from two MSc theses (Giraudo, 2015; Rodriguez-Villamil, 2015) on Scotch malt whisky which Nazli Koseoglu supervised during her PhD. Prof. Kate Heal and Prof. Dominic Moran provided feedback on the structure and content of the chapter drafts and contributed to the editing of the manuscript.

Chapter 4 Water use in the Scotch whisky industry

Abstract: Scotch whisky brings in the highest export profits to Scotland after oil and gas. Its production requires large volumes of high quality water and can only take place in Scotland according to legislation. Despite its economic significance as well as its dependency on local resources as a water intensive product, little research is available on the water use in the Scotch whisky supply chain and its valuation. In this study the water in the supply chain of Scotch whisky in analysed with bottom-up water footprint (WF) methodology and the economic value of water use in the industry is estimated using the marginal productivity approach. The results of the water footprint analysis highlight the advantage of the wet Scottish climate and the environmental impact of fertiliser used in barley production. 95% of the water footprint of Scotch whisky production chain comes from its supply chain, mainly water required to grow barley and assimilate the resulting pollution. Marginal productivity analysis estimated £ 5.6 /m^3 average value for water use in distilleries. Although the share of manufacturing processes is less than 5% of the Scotch whisky water footprint, the highest value is created in these processes.

4.1 Introduction

The whisky industry is significant to Scotland in terms of export volumes, economic value and cultural identity (Glenk et al., 2012). It is one of the most important manufacturing industries in the UK, creating 35,000 jobs across Scotland and generating £3.45 billion in exports as Scotland's second largest export, after oil and gas (SWA, 2011). 1.16 million bottles of Scotch whisky were exported in 2015 alone (SWA, 2015a). The industry's total contribution to the UK economy in 2013 was over £5 billion (SWA and 4-Consulting, 2015). The industry also contributes to the rural economy through its strong links with tourism. The total annual turnover from whisky tourism is estimated to be nearly £80 million (4-Consulting and SWA, 2011). Whisky production uses a significant volume of freshwater. With malting barley, water is one of its two main ingredients creating economic value as a result of the high demand and market price of the final product.

Increasing global demand for whisky (SWA and 4-Consulting, 2015) and projections for reduced seasonal water availability in several regions of Scotland (Brown et al., 2012) suggest water related pressures on the industry. The quantity and volume of water embedded in the supply chain of whisky and its value to the industry have to be estimated in order to understand the value of the water rights allocated to this industry. The research addresses this gap by providing results that estimate the water use and its value in the whisky industry and informs pricing and re-allocation decisions at catchment level by policymakers.

The Water Footprinting (WF) approach enables to account for water use in terms of direct water consumption in production processes and the supply chain as well as pollution resulting from both. The WF method is also useful because it produces volumetric figures per unit of output, in the case of whisky litres of pure alcohol (LPA), which in turn allows comparison with previous studies of the beverage industry. For the valuation part, the marginal productivity approach (MPA) is adopted to estimate the £/m^3 value of water use in the manufacturing processes of Scotch whisky. MPA is chosen as the methodology as it performs well under limited data availability and has been used in similar valuation studies of water use in manufacturing industries previously (Wang and Lall, 2002; Strzepek et al., 2006; Ku and Yoo, 2011; Nahman and DeLange, 2012; Tobarra-Gonzalez, 2015).

Section 2 provides an overview of the current literature in water footprinting methodology in food and beverage sectors and valuation of water in manufacturing industries. Section 3 looks into water use in Scotch whisky industry. It then sets out the methodology of the WF analysis of the whisky production system and MPA of water use in Scottish distilleries. Section 4 outlines data sources used in the analyses. Section 5 presents and discusses the findings of both analyses. Section 6 presents conclusions.

4.2 Literature

Primary agricultural production, as well as food and beverage industries, have received a lot of attention in the WF literature as a result of global food supply chains and intensive water demand from the agricultural sector.

Dairy farms and milk production systems (Ledgard, 2012; Zonderland-Thomassen et al., 2014; Palhares and Pezzopane, 2015), aquaculture (Mungkung et al., 2013; Auchterlonie et al., 2014), and tea (Jefferies et al., 2012) are some of the primary agricultural products previously analysed. In addition, the WF were analysed for food products manufactured from agricultural products, such as pork, poultry, livestock based products (Chapagain and Hoekstra, 2003; Ercin et al., 2012; Gerbens-Leenes, 2013), olive oil (Salmoral et al., 2011), soya products (Ercin et al., 2012), pasta (Ruini et al., 2013), chocolate (Ridoutt et al., 2009), as well as of a range of beverages (SAB Miller and WWF 2009; SABMiller et al., 2010;

Coca-Cola Europe 2011; Herath et al., 2013; Ene et al., 2013; Christ 2014; Lamastra et al., 2014).

WF methodologies are classified under three main categories: bottom-up, top-down and hybrid water footprints. While bottom-up WF are based either on life cycle assessment (LCA) or input-output assessment (IOA), hybrid approaches combine LCA and IOA with the bottom-up methodology created by the Water Framework Network (WFN) (Hoekstra et al., 2011). The bottom-up approach has been the most widely used in assessments of industrial food products because it facilitates more systematic and detailed analysis as well as more precise results for agriculture-based products. The bottom-up approach tends to result in higher WF estimates for same data set compared top-down approach as it captures the direct water use, specifically in agricultural inputs or products, better (Feng et al., 2011).

The economic valuation literature estimates the valuation of water use in manufacturing industries under three main types of function: demand, cost and production. Demand and cost functions require detailed information about water use structure (mains or private supply, percentage of recirculation) and pricing of each input (Rees, 1969; Turnovsky, 1969). While the demand function has not been widely used, the cost function has been a popular approach in literature to date despite the lack of competitive market conditions it assumes (Grebenstein and Field, 1979; Babin et al., 1982; Ziegler and Bell, 1984; Renzetti, 1992; Renzetti, 1993; Dupont and Renzetti, 2001; Reynaud, 2003; José and Reynaud, 2005; Kumar, 2006; Reynaud and Thomas, 2013).

The production function approach is based on the premise that firms maximise output for a given set of inputs. The cost function is linked to the production function (Fuss and McFadden, 1978) if the technology remains constant. The equivalence of the production and cost functions originates from the assumption that marginal costs should be equal to marginal values when firms compete in perfectly competitive markets (Wang and Lall, 2002), which rarely exist in reality.

The production function can estimate either average or marginal contribution of inputs to production and output elasticities. Users' maximum willingness to pay for water (SNIFFER, 2004) can be estimated through marginal productivity assuming that firms would be willing to pay only as much extra for a specific input as this input adds to the value, or market price, of the output. Thus, what firms actually pay and their valuation of the input can be compared to assess current pricing policy and future pricing strategy (Nahman and DeLange, 2012) and further re-allocation decisions. The marginal productivity method has so far been employed in studies looking at the value of water to manufacturing industries in different countries (Wang and Lall, 2002; Strzepek et al., 2006; Ku and Yoo, 2011; Nahman and DeLange, 2012; Tobarra-Gonzalez, 2015).

4.3 Methodology

There are two categories of Scotch whisky: single malt and single grain. Single malt whisky is classified as whisky produced from only water and malted barley at a single distillery, though not necessarily the product of a single batch or matured in a single barrel, while a blended single malt Scotch whisky is composed of single malt whiskies distilled at more than one distillery. Whilst single grain whisky is also distilled at a single distillery, it is made from grains, such as maize, wheat or rye, in addition to malted barley. In this analysis, we focus on the single malt industry whose production system is standardised as a result of legislation.

In recent decades, the United Kingdom and the EU have reshaped existing regulations as well as making new ones to protect the authenticity of products associated with a certain region through certifications such as Protected Geographical Indication. The legislative interest in the whisky industry started with the UK Scotch Whisky Act (1988) and the European Council Regulation- Annex II on spirit drinks (2008). While several regulations are relevant to the Scotch whisky, the main legislation that regulates, defines and enforces standards is the UK Scotch Whisky Regulation (2009).

Water and malting barley are the two main ingredients of single malt Scotch whisky. Whisky supply chain begins with spring barley cultivation. Barley, not necessarily harvested in Scotland, that meets certain malting standards set in Scotch Whisky Regulation (UK Parliament, 2009) is delivered to the malting plant for malting process. The barley grain is kept in a warm and moist environment optimal for germination. The spouting grain is dried, mashed, mixed with and stepped in large quantities of hot water to convert its starch content to sugar and dissolve the sugar in water. The resulting liquid is called wort. Yeast is added to the wort before it is left to ferment. The fermented malt is brought to the distillery for the pot still distillation process where the final product of whisky is distilled. The new spirit is first diluted with water then transferred into oak casks and stored in warehouses to mature for a minimum of 3 years. The alcohol content of the whisky stored in the cask may increase during this time period up to 60%. Before it is bottled and packed to be sold into market, it is diluted to its "bottling strength" which is between 40% and 46% alcohol content (Arnison and Carrick, 2015). There are also distilleries that store whiskies without any dilution to reduce inventory costs (Buxton and Hughes, 2014).

The production process includes several water intensive steps with different environmental impacts and economic values. In the analyses of water use, the production system is divided into two: supply chain and production chain. Manufacturing processes (direct WF in production chain) and procurement of raw materials (indirect WF in the supply chain) used in the manufacturing processes are assessed in a bottom-up water footprint analysis, while the valuation analyses considers solely the manufacturing aspect of the whisky industry that takes part in the distilleries. The processes that occur in the malting house as part of manufacturing Scotch whisky are not included in the valuation analysis due to data constraints. Supply chain processes beyond packing are not in the scope of the WF study.

4.3.1 Water footprinting analysis

The water footprinting (WF) methodology systematically analyses the water embedded in the production system through three distinct categories of water footprint, each of which have different impacts on water resources: green water footprint (WF_{green}), blue water footprint (WF_{blue}) and grey water footprint (WF_{grey}) (Hoekstra et al., 2011) as shown in Equation. 4.1:

$$\mathrm{WF_{total}} = \mathrm{WF_{green}} + \mathrm{WF_{blue}} + \mathrm{WF_{grey}} \tag{4.1}$$

The WF_{green} is water stored in soil moisture and lost through evapotranspiration. This is an indicator of the amount of precipitation taken up by plants and evaporated from soil as a result of crop cultivation activities. WF_{blue} refers to the consumptive use of available surface and groundwater resources. Its ready availability to all water users makes it flexible in allocation, and thus more valuable. Input material, evaporation and process water are examples of WF_{blue} in production processes (Milà i Canals et al., 2009). WF_{grey} indicates freshwater pollution resulting from an activity in volumetric terms of freshwater required to assimilate a pollutant load in a receiving water body (Hoekstra, 2015) and translates resulting pollution into volumetric terms comparable to other WFs. Figure 4.1 summarises material flows and different colour-coded footprints of water use for the production system of single malt Scotch whisky.

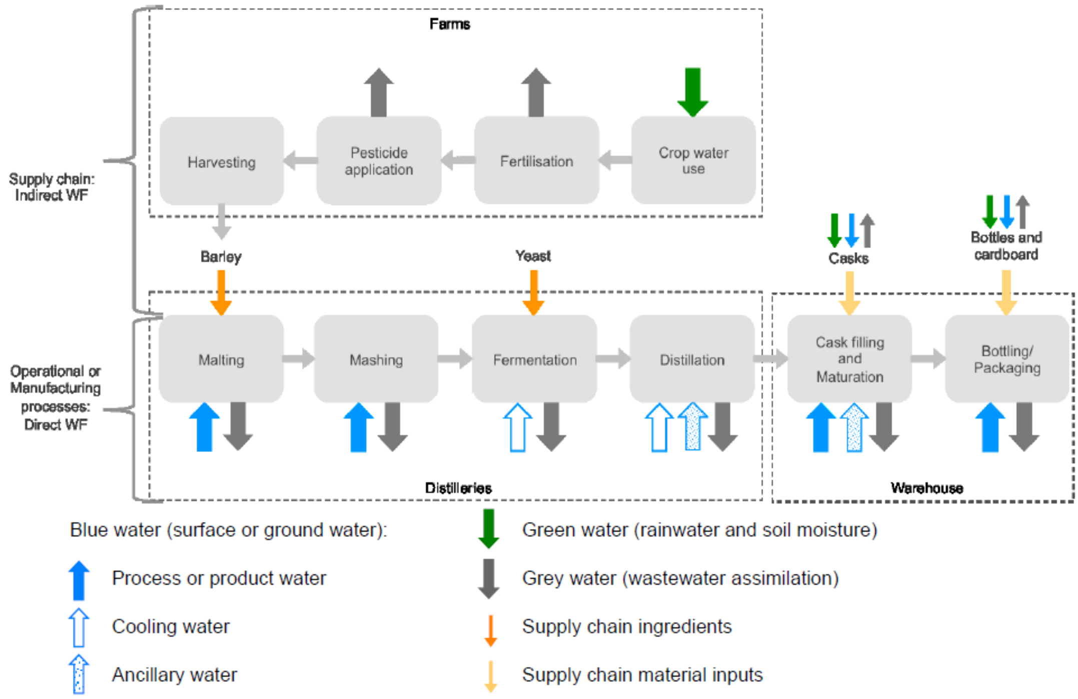

Figure 4.1. Supply and production chain of Scotch whisky (Rodriguez-Villamil, 2015).

The WF calculations involve two stages. The first stage calculates the indirect WF, resulting from provision of raw materials. The second stage addresses the direct WF, which originates from manufacturing processes. The results from both analyses are summed up to obtain the total WF of the entire production system.

4.3.1.1 Indirect WF

The indirect whisky WF comes from three materials: barley used in distilling, cardboard and glass bottles both used in packaging. Barley is the most important component of the whisky supply chain together with water. The total crop water requirement, effective rainfall and irrigation requirements are estimated by using the CROPWAT model (Allan et al., 1998; FAO, 2009).

Crop evapotranspiration (ET_c) (Equation 4.2) is estimated based on the reference evapotranspiration (ET_o) and crop factors using FAO's Penman-Monteith climatic data approach (Allan et al., 1998). ET_c for barley is estimated using the 'irrigation schedule option' and 'no irrigation (rainfed)' choice as these approaches balance soil moisture over the growing period of the crop (Rodriguez-Villamil, 2015). Effective rainfall is also calculated with CROPWAT using the USDA soil conservation method (USDA, 1968).

$$ET_{green} = ET_c = \Sigma ET_a \tag{4.2}$$

where,

ET_c= crop evapotranspiration (mm)

ET_{green}= total green water evapotranspiration (mm) over the growing period

ET_a= adjusted evapotranspiration per day through the growing season (mm/day)

Barley and cereals are usually grown under rain-fed conditions in Scotland (Steduto et al., 2012). For this reason, blue water use in irrigation (ET_{blue} = 0) is not considered (Rodriguez-Villamil, 2015).WF_{green} for barley is estimated based on water use of crops in a given year and area (Equation 4.3).

$$WF_{green,barley} = \frac{CWU_{green}}{Y} = \frac{10 \times ET_{green}}{Y} \tag{4.3}$$

where;

CWU_{green}= crop water use (m^3/ha)

Y= the yield for spring barley (tonne/ha) in the different agricultural regions

While some distilleries use only Scottish barley, some may import from England. 90% of the spring barley used in distilling is grown in the UK (Flaviar, 2016). In the analysis, it is was assumed that all barley production takes places on the east coast of Scotland. Productivity, climate, soil and regional production characteristics of 7 most representative agricultural sub-regions accounting for over 90% of the barley production in Scotland (Scottish Government, 2015a) are used in estimation of average yield for spring barley (Rodriguez-Villamil, 2015).

Nitrogen and phosphorus in fertilisers are the pollutants taken into consideration based on availability of data and their significant contribution to diffuse pollution and eutrophication (Liu et al., 2012). The $WF_{grey,barley}$ is estimated separately for both pollutants and the largest figure is used as the critical pollutant for final WF_{grey}. The volume of water required to assimilate the critical pollutant is assumed to be sufficient to assimilate the other. The WF_{grey} associated with production of spring barley was estimated with Equation 4.4.

$$WF_{grey,barley} = \frac{\frac{L}{(C_{max} - C_{nat})}}{Y_{fertiliser}} = \frac{(\alpha \times AR)/(C_{max} - C_{nat})}{Y_{fertiliser}} \tag{4.4}$$

where:

L= the pollutant load (kg/ha)

$Y_{fertiliser}$= the yield of spring barley (tonne) per area (ha) of fertiliser application

C_{max}= maximum acceptable pollutant concentration (mg/l) indicated in the regulations for human consumption

C_{nat}= natural background concentration of the pollution in the receiving water body

α = leaching run-off fraction (%) of the applied chemical that reaches freshwater

AR= application rate of fertiliser (kg/ha).

The WF for packaging is estimated by considering the WF of glass bottles and paperboard (Rodriguez-Villamil, 2015). The WF of glass bottles is assessed based on total water use in the life-cycle assessment of glass (Blackett, 2012), average weight and number of 70cl bottles required to bottle LPA produced per year, assuming one LPA is equivalent of 3.56 bottles (SWA, 2015b), and an average recycling rate of glass in Europe (54%) without reuse. The WF of paperboard is estimated according to the average weight of packaging required for paperboard used in whisky based on previous WF figures for solid bleached paperboard sourced from an unspecified location (van Oel et al., 2009; Jefferies et al., 2012; Rodriguez-Villamil, 2015).

4.3.1.2 Direct WF

Direct WF considers the WF_{blue} in process steps and used as evaporative water and the WF_{grey} resulting from spirit production ($WF_{grey,spirit}$) as there is no green water use in the manufacturing stages of production. The estimation for $WF_{blue,spirit}$ (m^3/year) is based on the chain-summation approach in which the WF of each processing step is added to calculate the final WF of production (Hoekstra et al., 2011). Estimations of steeping/malting, mashing, strength reduction and cooling water are made using average water use, compiled from the literature (Table 4.1). Total $WF_{blue,spirit}$ resulting from four production processes was calculated as the sum of the process steps divided by LPA. Water lost through evaporation was calculated in terms of water vapour in the malting process and in spirit evaporation during maturation (Rodriguez-Villamil, 2015).

Table 4.1. Process water use for each process step for malt whisky production considered in this study (compiled from (Rodriguez-Villamil, 2015).

Process step	Process water use	Water source	Source(s)
Steeping water	4 m^3/tonne of barley	Private supplies of groundwater	(Briggs, 1998; Eureka Swan, 2002; Russell et al., 2003; SNIFFER, 2004; Black et al., 2006; Broadbent and Ham, 2013)
Mashing water	9.22 m^3/tonne malt	Private supplies of groundwater	(Lea and Piggott, 1995; Russell et al., 2003; Bringhurst and Brosnan, 2014)
Strengthreduction water	-	Potable water (mains supply)	(Rodriguez, 2015)
Cooling water	120 m^3/tonne malt	Surface water (streams)	(Russel et al., 2003; SNIFFER, 2004; SEPA, 2005)

It is assumed that each malt distillery has its own malting site, dark grains plant and biological treatment plant. Pot ale, spent wash or effluent from the first distillation is transformed into pot ale syrup which can be used for cattle or pig feed, or together with draff, the malt mashing residue, can be made into barley dark grains for cattle and horse feed. The wastewater from animal feed production, the spent lees, effluent from the second distillation, and effluent from the malting process are then treated in the biological treatment plant (Russell et al., 2003) before the final liquid residue is discharged to surface waters or sewage (Rodriguez-Villamil, 2015).

Pollutants in the distilling effluents for malt whisky distilleries originate from either distilling or cooling processes. While distilling effluents contain copper, zinc, lead, ammonia, suspended solids (SS), biological oxygen demand (BOD) and have a low (acidic) pH, cooling effluents may cause thermal pollution (SEPA, 2016b). The effluents of whisky distilling mostly consist of biodegradable components with the exception of significant copper residues from distillation stills found in pot ale and spent lees effluents. Copper stills are an essential part of the distilling process and significant volumes of copper residues that can cause potential harm in the receiving water bodies might be found in the effluents. Therefore, copper is considered as the most critical pollutant amongst the non-organic pollutants associated with whisky production (Russell et al., 2003).

In the scope of the analysis here $WF_{grey,spirit}$ is broken into its major components as shown in Equation 4.5 (Rodriguez-Villamil, 2015).

$$WF_{grey,spirit} = WF_{grey,BOD} + WF_{grey,copper} \tag{4.5}$$

where:

$WF_{grey,BOD}$: the oxygen required to degrade organic components in the effluent

$WF_{grey,copper}$: amount of water required to dilute maximum allowable discharge concentration to natural concentration in the receiving water body.

Both of which are calculated separately according to Equation 4.6.

$$WF_{grey,spirit} = \frac{L}{(C_{max} - C_{nat})} = \frac{(V_{effl} \times C_{effl}) - (V_{abs} \times C_{act})}{(C_{max} - C_{nat})} \tag{4.6}$$

where:

L: the pollutant load expressed by the amount of pollutant in the effluent (μg)

C_{max}: CAR licence for the Glenallachie Distillery in Aberlour, Speyside (μg/l), distillery chosen on the basis of data availability

C_{nat}: Natural copper concentration (μg/l)

C_{effl}: Copper concentration in treated effluent (μg/l)

V_{effl}: Volume of the effluent (l)

V_{abs}: Volume of the receiving water body (l)

C_{act}: Actual copper concentration in the receiving water body (μg/l)

Other important sources of pollution that contribute to $WF_{grey,spirit}$ but are not accounted for here are the manufacture of yeast and the water required to wash fermenters, process pipes, filtration systems and other ancillary water uses (Rodriguez-Villamil, 2015).

4.3.2 Marginal productivity approach

Here the MPA approach is adopted to analyse water value in the production of Scotch malt whisky due to the previously mentioned limitations regarding the availability of price data and lack of competitive markets that determine water allocation in Scotland.

The mathematical representation of the production function, from which marginal productivity is derived, relates to input and output quantities (Equation 4.7 and 4.8):

$$Q = f(K, L, W, E, M) \tag{4.7}$$

$$Q = \mathrm{A}\, \mathrm{K}^{\alpha_{\mathrm{K}}} \mathrm{L}^{\alpha_{\mathrm{L}}} \mathrm{W}^{\alpha_{\mathrm{W}}} \mathrm{E}^{\alpha_{\mathrm{E}}} \mathrm{M}^{\alpha_{\mathrm{M}}} \tag{4.8}$$

where:
Q: the production or output
A: total factor productivity, the portion of output not explained by the amount of inputs used in production
K: capital
L: labour
W: water
E: energy (electricity)
M: raw materials, here mainly malting barley
α_K: output elasticity of capital
α_L: output elasticity of labour
α_W: output elasticity of water
α_E: output elasticity of energy
α_W: output elasticity of raw materials

Output elasticity, the percentage change of output volume/mass (Q) of a firm divided by the percentage change of an input, in this case water (W), is estimated with Equation 4.9.

$$\alpha_W = \frac{\partial_Q/Q}{\partial_W/W} \tag{4.9}$$

where:
∂_Q: change in the amount of output,
∂_w: change in the input of water,

From the elasticity of production function, marginal productivity can be estimated by how much a unit change in water use would change the output, with all other inputs remaining constant (Equation 4.10).

$$\rho_W = \frac{\partial_Q}{\partial_W} = \alpha_W \cdot \frac{Q}{W} \tag{4.10}$$

where:

ρ_W: marginal productivity of water

4.3.2.1 Valuation of water use in the Scotch industry

Following previous MP analyses, initially a production function was estimated using functional forms of the Cobb-Douglas (CD) (Equation 4.11) and Trans-log, a more generalised version of CD (Equation 4.12) where parameters are as defined in Equation 4.7.

$$In\,Q = In\,A + \alpha_K\,InK + \alpha_L In\,L + \alpha_E\,In\,E + \alpha_W In\,W + \alpha_M In\,M \tag{4.11}$$

$$\begin{aligned} In\,Q = \;& \alpha_o + \alpha_K\,InK + \alpha_L In\,L + \alpha_E\,In\,E + \alpha_M In\,M + \alpha_W In\,W + \\ & \tfrac{1}{2}\alpha_{KK}\,InK^2 + \alpha_{KL}\,InKInL + \alpha_{KM}\,InKInM + \alpha_{KE}\,InKInE + \\ & \alpha_{KM}\,InKInM + \alpha_{KW}\,InKInW + \tfrac{1}{2}\alpha_{LL}\,InL^2 + \alpha_{LM}\,InLInM + \\ & \alpha_{LE}\,InLInE + \alpha_{LW}\,InLInW + \tfrac{1}{2}\alpha_{MM}\,InM^2 + \alpha_{ME}\,InMInE + \\ & \alpha_{MW}\,InMInW + \tfrac{1}{2}\alpha_{EE}\,InE^2 + \alpha_{EW}\,InEInW + \tfrac{1}{2}\alpha_{WW}\,InW^2 \end{aligned} \tag{4.12}$$

However, these approaches did not describe the structure of the data sets. A Trans-log functional form exacerbates the multi-collinearity already present in the data set and a CD assumes perfect substitution between production factors which does not apply to the whisky case. As explained further in Appendix E, these functions are not used in the final estimation due to these limitations of the data set. Therefore it was decided to estimate a model using a mixed functional form, the CD-Leontief production function (Equation 4.13). A Leontief function of fixed proportions enabled the treatment of water, energy and material factors as a single variable in model assuming there is no perfect substitutability between these factors. With this assumption, it was possible to deal with the high collinearity in the data set and assume lack of substitution between collinear production inputs (W,E,M) (Giraudo, 2015).

$$Q = F(c(L,K),\ l(\min(W, E, M)) \tag{4.13}$$

where:
F (c (L,K): CD part of the function
L (min (W, E, M)): Leontief part of the function

Equation 4.13 can be further broken into three parts as indicated in Equations 4.13, 4.14 and 4.15 for each of the three inputs of W, E and M that is highly correlated with other two.

$$In\ Q = In\ A + \ \alpha_K In\ K + +\alpha_L In\ L + \alpha_W In\ W \quad (4.14)$$

$$In\ Q = In\ A + \alpha_K In\ K + +\alpha_L In\ L + \alpha_E In\ E \quad (4.12)$$

$$In\ Q = In\ A + \ \alpha_L In\ L + \alpha_L In\ L + \alpha_M In\ M \quad (4.13)$$

4.4 Data

Data for the WF estimations were collected from information published in official government reports and other relevant literature on Scotland where available. Where no domestic information is available, international literature and databases are used (Table 4.2).

Table 4.2. Data used in Scotch whisky WF analysis (compiled from (Rodriguez-Villamil, 2015).

Indirect $WF_{green,barley}$	**Source(s)**
Climate	UK Climate Projections and database (UK Met Office, 2009; 2014)
Soil and crop growing areas	National Soils Inventory for Scotland (Lilly et al., 2011), barley production in UK (HGCA, 2005; 2012: 2013)
	and CROPWAT model (FAO, 2009)
Crop water requirement	(Allan et al., 1998; Chapagain and Hoekstra, 2004; HGCA, 2005; FAO, 2009; Steduto et al., 2012)
Average spring barley production (tonnes, harvest area and yield)	Economic Report on Scottish Agriculture 2015 (Scottish Government, 2015a)
$WF_{grey,barley}$	**Source**
Fertiliser application rate and area applied (%) for 2011-2013	(Defra, 2015)
Average fertiliser leaching-runoff fractions	(Franke et al., 2013)
Maximum allowable concentrations	(European Commission, 1991a)*
Paper cardboard	(van Oelet al., 2009; Jefferies et al., 2012)
Glass bottles	(ECGF, 2010)

Direct WF_{blue}	Amount	Source(s)
Steeping water	4 m^3/tonne barley	(Briggs, 1998; Eureka Swan, 2002; Russell et al., 2003; SNIFFER, 2004; Black et al., 2006; Broadbent and Ham, 2013)
Mashing water	9.22 m^3/tonne malt	(Lea and Piggott, 1995; Russell et al., 2003; Bringhurst and Brosnan, 2014)
Strength reduction water	-	Own calculations (Rodriguez-Villamil, 2015)
Cooling water	120 m^3/tonne malt	(Russell et al., 2003; SNIFFER, 2004; SEPA, 2005a)
Direct $WF_{grey,copper}$	**Amount**	**Source(s)**
Copper concentration in untreated andtreated spent lees	25-40 mg/l 1.5 mg/l	(Murphy et al., 2009)
EQS** for copper in freshwater	1 µg/l	(SEPA, 2016b)
CAR licence for the Glenallachie Distillery in Aberlour	330 µg/l	(Robinson, 2015)
Actual copper concentration in representative river in Scotland	2.82 µg/l	(UNEP, 2009)
$WF_{grey,BOD}$	**Amount**	**Source(s)**
Application rate of Total nitrogen (N)	105.33 kg/ha	(Defra, 2015)
Application rate of Total phosphorus (P)	53.67 kg/ha	(Defra, 2015)
BOD*** concentration in effluent	20 mg/l	(Russell et al., 2003)
EQS for BOD in freshwater	7-19 units	(SEPA, 2016b)
Council Directive BOD discharge limit	25 mg O_2/l	(European Commission, 1991b)

Natural BOD concentration	0 mg/l	(EEC, 1978)
Actual BOD concentration in representative rivers in Scotland	2.82 mg/l	(UNEP, 2009; Blair, 2015)

*Natural concentration of the fertiliser constituent components, e.g. phosphate and nitrate in the receiving freshwater assumed to be insignificant, **EQS (Environmental Quality Standard), *** Biological oxygen demand

Most of the data input required for MP analysis was developed based on a set of assumptions, literature and a limited number of publically available sources of information due to competition rules in the industry. Instead of 169 distilleries in Scotland, the analysis here is based on only 43 members of the Scottish Whisky Association environmental scheme that disclosed additional information on their resource use in the Annual Report of SWA (2015). Table 4.3 summarises the sources of information used in the analysis and Table 4.4 provides statistical descriptions of data for each production input.

Table 4.3. Variables for MP analysis (compiled from Giraudo, 2015).

Variable	Source(s)
Total revenue, Q	Average price per litres of pure alcohol (Gary, 2014), volume of LPA produced from 43 distilleries where data is available (Giraudo, 2015**)**
Labour, L	Number of people employed by distilleries (Robinson, 2015), surveying distilleries and expert opinion to estimate ratio of people directly involved in the distillation process among total number of employees
Value of capital assets, K	(1) land: size of land (Robinson, 2015) and surveying distilleries; average market value of land per acre (Savills, 2015) and by type (Scottish Government, 2015b),
	(2) buildings: the average cost of distillery buildings per metre square (SSA 2005; 2013)
	(3) water, building services and electricity infrastructure (Engineering Company, 2015)*
Energy, E	The average cost of energy (Gary, 2014), energy price index energy mix used across the industry (SWA, 2013) average price of different sources of energy (gas, electricity and heavy fuel) for industrial firms (UK Government, 2015)
Materials, M	Based on water intake and production function of pure alcohol and whisky (Buxton and Hughes, 2014) and material use efficiency assumptions for distilleries of different age and renovation year (Ronde 2013; Ronde 2014; Anonymous Contact 2015)*
Water, W	Abstraction licence registry (SEPA, 2015) and their average use ratio (Daalmans, 2015).
Technology	Age of the distillery used as a factor (Brown et al., 2012; Russell and Stewart, 2014; Hughes, 2015) and water efficiency of the distillery is based on whether it is built or renovated before or after Water Framework Directive (European Commission, 2000).

*****Several individuals and the consulting company that provided information requested to remain anonymous.

Table 4.4. Statistical description of data inputs for marginal productivity analysis from 43 distilleries in Scotland (adapted from Giraudo, 2015).

Variable, unit*	Minimum	Maximum	Mean	SD**
Yield (Y), LPA	211500	20727000	6262859	4081627
Capital (K), £	3002590	224560016	12695417	33236274
Labour (L), person	4	77	17.98	13.916
Materials (M), barley in tonne	11	2522	718	594
Energy (E), electricity in kWh	237076	58933040	15575771	13448874
Water (W), m^3	10000	2386400	707494	567381

*These units are monetised using UK 2015 market values, ** SD: Standard Deviation

4.5 Results and Discussion

This analysis provides an estimate of the occurrence of water use, related environmental impact and where the most value from water use is created in the production system of Scotch whisky. The results estimated here indicate that while water is mostly used in barley production, either in the form of soil moisture taken up by the crops or a dilution water for diffuse pollution caused by the fertilisers, the main value in the production system of whisky is created in the manufacturing processes that take place in the distilleries.

4.5.1 WF analysis

The WF of Scotch malt whisky was estimated based on the 275 MLPA (million LPA) produced from 835,000 tonnes of malting barley used in the whisky industry in Scotland in 2013 (SWA, 2014a; 2014b). The total WF was estimated by considering green water ($WF_{green,barley}$ and $WF_{green,paperboard}$), consumptive blue water ($WF_{blue,spirit}$, $WF_{blue,bottles}$ and $WF_{blue,paperboard}$) and grey water ($WF_{grey,barley}$ and $WF_{grey,spirit}$).

Results illustrate that water is mainly embedded in the supply chain and the total WF of the whisky industry comes from green water used in barley production. Barley production relies solely on WF_{green} in Scotland. WF can only be used by crops, thus the only opportunity cost is choosing to grow barley among available crop options and currently this is not an issue, making it the WF with lowest environmental impact and opportunity cost. 96.4% of the Scotch whisky industry's total WF originates from the production of barley for malting, which highlights the advantage of a wet climate in rain-fed agriculture. However, future climate projections show that there will be higher temperatures and less rainfall during the summers in the barley growing regions (Brown et al., 2012; HGCA, 2013). This also means higher competition among water users, including barley irrigation, which will increase due to decreased green water availability for crops (Brown et al., 2012).

Here blue water withdrawal is considered solely in the operational processes that take place in distilleries and is quite insignificant (2%). Grey water mainly originates from fertiliser diffuse pollution and accounts for almost all of the grey WF for malt whisky production. Estimation of WF_{grey} is particularly relevant to water policy as preventing eutrophication and diffuse pollution from agricultural run-off and reducing pollution from effluents containing copper and BOD would contribute greatly to water quality improvement and pollution control costs at catchment level. In addition to physical water scarcity concerns, high grey water footprint (28%) caused by fertiliser use in barley production (26%) and BOD and copper rich effluents from the distilleries (2%) introduces an economic scarcity of water. The pollution increases treatment costs of water to all its users, including the distilleries themselves. The whisky industry can financially contribute to ecosystem schemes and catchment initiatives in collaboration with other stakeholders and contribute to water quality improvements in the catchments in which they source their barley or where the distilleries are based.

Thermal pollution from distillery effluents may also be a source of WF_{grey}. However, it is not part of the analyses here due to insufficient data on effluent temperatures and the possible reuse of cooling water in production processes. Thus, it is assumed that the temperature increase is within the limits after recirculation or reuse in heat exchange systems (Daalmans, 2015; SEPA, 2011).

The results of previous beverage WF studies together with results from Rodriguez-Villamil (2015) in terms of total, direct and indirect WF are summarised in Table 4.5. Findings from previous beverage studies might not be fully transferable to Scotch whisky. Among these studies, only Italian wine (Lamastra et al., 2014) is a geographically protected product. Geographically protected products indicate a greater reliance on local (water) resources as the production of the beverage can take place only in a certain region and relocating the industry or outsourcing production is not a possibility. None of the products analysed are a distilled alcoholic beverage, which requires significant volumes of water for the dilution of pure litres of alcohol at the end of the processes, unlike fermented alcohols such as beer or wine or non-alcoholic beverages e.g. Coca-Cola. However, an obvious trend across all studies is that a significant amount of WF lies in the production of the raw ingredients (grapes, barley or sugar). For this reason, production systems based on rain-fed and local crops are expected to have lower WF compared to others that produce the same end product (SABMiller and WWF, 2009; SABMiller et al., 2010).

Table 4.5. Previous WF analyses in the beverage industry.

Citation	Case study	Location(s)	Total WF	Indirect WF	Direct WF
Rodriguez-Villamil, 2015	Scotch whisky	Scotland	635	612	23
Ene et al., 2013	Medium scale winery	Romania	1844 l/l wine	99.9%	0.1%
Herath et al., 2013	Wine	New Zealand	823l/FU*	Type of methodology used interpret these components differently	
Lamastra et al., 2014	Wine	Italy	697** l/wine	91*%	9%
SAB Miller and WWF 2009***	SABMiller breweries	South Africa	155 l/l beer	93.8%	1.7%
		Czech Republic	45	91.7%	8.3%
SABMiller et al., 2010***	SABMiller breweries	Peru	61 l/l of beer	90%	10%
		Tanzania	180	92%	8%
		Ukraine	166	92%	7%
Chapagain and Hoekstra 2007	Coffee	The Netherlands (manufacturing)	140 l/cup	-	-
	Tea		34		
Coca-Cola Europe 2011	Coca-Cola	10 European regions	35 l/l coke	99%	1%

* FU: functional unit defined as a 750-ml bottle of wine at the winery gate. **Average of values among 6 vineyards analysed in Sicily, Italy.***SABMiller analyses include bottling also in the operational WF

4.5.2 MP Analysis

The mixed CD-Leontief function using Ordinary Least Squares (OLS) regression estimation provides a good fit indicated by the adjusted R-square statistic (0.75) and the signs of the regression coefficients detailed in Appendix E. All coefficients except capital are statistically significant (as $p < 0.01$). The statistical insignificance of capital can be explained by certain degrees of randomness in the value of capital assets introduced by the approach used in the estimation of land and building values. This randomness also yields a low correlation between revenue and capital (0.031) though both data sets were constructed using production capacity as a proxy (Appendix E). The elasticity and valuation results from the mixed CD-Leontief function are reported in Table 4.6.

Table 4.6. Marginal value and elasticity (Giraudo, 2015).

Average	**Marginal value of water**	**Output elasticity of water**
Mean	8.36 £/m^3	0.56%.
Median	5.56 £/m^3	

The results are estimated by multiplying the output elasticity for the 43 distilleries in the dataset by the average LPA assessed for each distillery. The distribution of marginal productivity values across distilleries sampled here is skewed (Figure 4.2) by outlier value observations as high as 32 £/m^3, thus the median of 5.60 £/m^3 is assumed to be more representative than the mean value of 8.40 £/m^3. Even with this assumption, the marginal value estimation of 5.60 £/m^3 for the water use in Scotch whisky production is high compared to value estimates for food and beverage industries from previous MP analyses (Table 4.7). The output elasticity result suggests that with 1% increase in water use, revenue increases by 0.56%, still a high estimate compared with the output elasticities in the beverage industry reported by Ku and Yoo (2011).

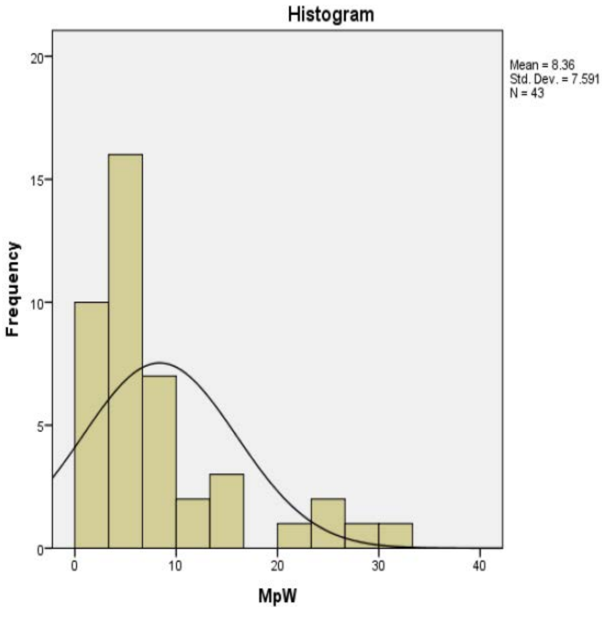

Figure 4.2. Distribution of marginal productivity values across distilleries (Giraudo, 2015).

Table 4.7. The study designs and main results from previous marginal productivity estimates in food and beverage industries.

Study	Region	Functional form	Estimation	Elasticity	Value (£/m^3) *
Wang and Lall, 2002	China	Trans-log	OLS	-1.04	0.8
Strzepek et al., 2006	World	Trans-log	SUR** IZEF***	-1.46	2.36
Nahman and Lange, 2012	South Africa	Trans-log	OLS	-0.78	0.02 and 1.0
Ku and Yoo,2011 ****	Korea	CD, Trans-log	Not given	0.0188 and 0.008	1.3 and 0.6
Tobarra-Gonzalez, 2015	Chile	Trans-log	Not given	-1.29	1.3

*Value observations from previous studies converted first by Purchasing Power Parity coefficients for each case study country (OECD, 2015) to international dollars, then to British Pound value of the study year (XE, 2015). Finally Pound values were converted to 2015 British Pound (Inflation rates 2015). **SUR: Seemingly unrelated Regressions,***IZEF: Iterative Zellner Efficient method, OLS: Ordinary Least Squares. **CD: Cobb-Douglas, ****Unlike other studies that report price elasticity, Ku and Yoo (2011) reports output elasticity of water use as we do here.

There are potentially two reasons for this wide difference between the results of this study and others reported in the literature. Firstly, whisky is a premium product compared to most other products of the beverage industry and previous MP studies looked at food and beverage industries in an aggregate manner rather than looking at a specific product. Secondly, the analysis here is not able to distinguish between water as an ingredient to whisky and water used as process water in production (or cooling). These two different water uses both take place in distilleries but have different values and elasticities. Water used in production processes can partially be substituted by technological improvements while water as raw material cannot be substituted by other inputs and can only be substituted with reduction in output as it is one of the two ingredients to produce malt whisky (UK Parliament, 2009).

It can also be assumed that the establishments (43 out of 169) that voluntarily provided information are at the more sustainable end of the industry. Therefore, the results might not be fully representative of the whole whisky industry in Scotland but of more environmentally aware distilleries and might be an overestimation of marginal productivity value and underestimation of the water footprint.

Table 4.8 summarises the results from two separate analyses of the WF and MP of the Scotch whisky industry, pairing the WF and the value estimate of water use for each step of the production system. The value estimates for parts of the production system beyond distillation included in Table 4.8 are transferred from previous analyses in order to provide a comparison of values and highlight where the most value is added to the water use in the Scotch whisky.

Table 4.8. Green, blue and grey water footprint (in ML water per ML of LPA) for each component and value estimate for each type of water use (partly compiled from (Giraudo, 2015; Rodriguez-Villamil, 2015).

WF component	Green WF	Blue WF	Grey WF	Total WF	% WF total	Value estimate	Valuation Method
Indirect WF	**442**	**3**	**167**	**612**	**96**	**-**	**-**
Spring barley	259	-	167	426	67	**0.3 £/m^3**	Value transfer (SNIFFER, 2004)
Glass bottles	-	1	-	1	0.2	3.6 £/m^3	Own calculation ***
Paper board	183	2	-	185	29	2 £/m^3	Own calculation ***
Direct WF	**-**	**11**	**11**	**23**	**4**	**5.6 £/m^3**	**MPA** (Giraudo, 2015)
Process water	-	10	11	22	4		
Steeping water	-	3	-	3	0.5		
Mashing water	-	7	-	7	0.1		
Strength reduction	-	0.1	-	0.1	0.02		
Cooling water**	-	80	-	80	13		
Evaporative water	-	0.5	-	0.5	0.08		
Total WF	**442**	**14**	**178**	**635**	**100**	**5.9 £/m^3**	

*All WF figures are estimated in million litres (Ml) of water per million litres of pure alcohol produced. **Cooling water is not considered in the final results due to lack of data availability. *** Own calculations are derived from a meta-analysis conducted in Chapter 3 that looks into the value of water. The values found in this analysis for paper and allied industries and all manufacturing industries are respectively transferred to paper board and glass used in the packaging.

4.6 Conclusions

Climate change requires changing the way we allocate and price water. Increasing competition and reduced availability necessitates safeguarding a supply for high value industries to achieve higher social returns from the use of available water. The whisky industry contributes significantly to the overall economy in Scotland through its use of Scottish water resources. WF and MP analyses presented here are conducted at a country level with the intention of producing an industry scale estimate of water use and its valuation to inform policy makers and the whisky industry of the current situation. WF analysis highlights water related risk along the supply chain (SABMiller and WWF, 2009; SABMiller et al., 2010; Coca-Cola Europe, 2011) of Scotch whisky and the MP approach estimates the maximum willingness to pay of the industry for its water use.

Although only 2% of the water use in terms of WF takes place in the distilleries, the distilleries rely on the quality and quantity of local water resources to maintain their operations. While Scotch Whisky Regulations allow the import of spring barley, the manufacturing processes have to take place in Scotland for any whisky to be classified as Scotch (UK Parliament, 2009). The high value of water use in distilleries (£5.6/m^3) indicates that distilleries would be willing to pay premium prices to safeguard their water supply when future competition makes water markets at catchment level a feasible option to reallocate available water in the most economically efficient way. However, the scale of the analyses here restricts the applicability of estimates to catchment level because the availability of water, and therefore its value and impact of its use, would be different in each whisky region, if not in each catchment in Scotland. Further research on constructing catchment level benchmarks and location-specific impact assessments that would account for different implications of water consumption on ecosystems would complement the findings here. Inclusion of the locality would also enable the consideration of opportunity costs of water for different production locations. Linking the WF figures with further valuation analyses by using water productivity estimations among different firms and regions (or catchments) (Chenoweth et al., 2014) to create industry and catchment level benchmarks for water use efficiency.

Chapter 5 Valuing water use in the Scottish livestock industry

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Intended for submission to *Agricultural Water Management*.

The candidate collected the data and performed analyses for allocation and valuation of different water uses and wrote the paper. Co-authors provided support and guidance on the scope and design of the project and contributed to the editing of the manuscript.

Chapter 5 Valuing water use in the Scottish livestock industry

Abstract: Although agriculture has a comparatively low contribution to GDP in developed countries, it is important in terms of supplying primary inputs to the food and drinks industry and for the rural economy. Although volumetric estimation of water use has received a lot of academic attention, the valuation of water use in the livestock industry has not. In our analysis, we estaimated the economic value of water use to Scottish livestock farmers using netback analysis. Value estimates for dairy and beef farms are similar, respectively £1.9 and £1.8/per m^3. These figures indicate that there is a gap between the actual value of water used on farms and what is being currently charged for it. Although the analysis uses data for Scotland, the results are transferable to other parts of the world with similar livestock systems and climate.

5.1 Introduction

Water is central to the livestock industry– it uses 8% of the global water supply, mainly for the irrigation of crops for livestock feed (Schlink et al., 2010). Water availability is expected to reduce as a result of climate change whilst the global demand for meat and dairy products is projected to increase (Forde, 2016; Worldwatch Institute, 2016). For this reason, the quantification of water use in the livestock industry and its effects on water resources have received substantial research attention internationally (Renault and Wallender, 2000; Galloway et al., 2003; Chapagain and Hoekstra, 2004; Steinfeld et al., 2006; Chatterton et al., 2010; Hanasaki et al., 2010; van Breugel et al., 2010; Herrero and Thornton, 2011; Peden, 2012; Zonderland-Thomassen et al., 2014; Murphy et al., 2017).

However, there remains the need to identify the gap between the real value of water to its users and actual prices and charges associated with its use on cattle farms in order to discuss economic efficiency and incentivise higher productivity of water use on farms.

When water is an intermediate good required in production processes, as it is in agriculture or in industry, then tariffs raise fewer equity concerns relative to household tariffs. Thus, it is reasonable to advocate the use of a pricing mechanism to manage water demand for commercial uses assuming that water is a good like any other without any "non-market" values when it is an input to production. Efficiency in commercial water uses is best achieved through tariff design that reflects user costs (Millerd, 1984). Indeed commercial water demand is directly linked with the demand for the final product and becomes a function of the price of water and the price of the final product that reaches the market.

The response of consumer demand to changes in pricing can be predicted using two economic concepts: willingness to pay and ability to pay for the supply of an additional unit of good and service (Joewono, 2009). Consumer willingness to pay (WTP) is the maximum amount that s/he would be willing to pay to access a service/good or to improve its current quality (Whitehead, 2006). It depends on the individuals' level of income and the perception of risk: the greater a person's aversion to risk losing this good or service, the more s/he will be willing to pay. On the other hand an individual's ability to pay (ATP) is the maximum amount that s/he is capable of paying and is therefore linked to income. In the case of water as an input to production, ability to pay is assumed to be greater than or equal to willingness to pay, even for users with a strong aversion to risk (Tabieh et al., 2015). Estimating livestock farmers' ATP for their water use is more representative than estimating their WTP in setting water charges that are both fair and affordable to farmers and yet simulate economically efficient use of water among all users and re-allocate water to the users who value it the most.

This study explores livestock farmers' ATP in the context of Scottish beef and dairy production by estimating the value of water use to the livestock farmers using netback analysis. The netback analysis method is used to determine the maximum amount farmers could pay for their water use while their businesses remain profitable. This method has previously been applied to the valuation of water use in agricultural irrigation (Bate and Dubourg, 1997; Tren, 1998; Lindgren, 1999; Nilsson et al., 2003; Esmaeili and Vazirzadeh, 2009) where water use dynamics are most similar to livestock. Therefore it is assumed that a similar approach is suitable for estimating the value of water use by livestock. Moreover, the data required for netback analysis of the livestock industry in Scotland can be compiled from publicly available sources. The ATP value per m^3 assessed here is expected to be transferable to other parts of the world with similar climatic conditions and farming practices.

The chapter is structured as follows. Section 2 provides background on livestock farming in Scotland. Section 3 considers the dynamics of water use on livestock farms and estimates how much water is used by a standard dairy or beef herd. Section 4 elaborates on data and the netback methodology used in the estimation of profit. Section 5 discusses the current cost of water use based on different supply options (mains water supply, private abstraction and rain harvesting) available on farms. Section 6 summarises and interprets results from the previous sections and Section 7 sets out the conclusions of the analyses.

5.2. Livestock farming in Scotland

Livestock production is a significant rural land use throughout Scotland. Scotland is one of the few countries in the EU with a rural economy and agriculture that relies heavily on livestock (Eurostat, 2012). Dairy and meat production contribute to almost one-third of food and drink manufacturing output as well as 2.5% of all manufacturing output of the Scottish economy in GDP terms (Ashworth, 2009). Table 5.1 shows the central role of livestock, particularly the cattle industry, in the agricultural economic output of Scotland.

Table 5.1. Contribution of the livestock industry to the annual economic output of Scottish agriculture (figures taken from Scottish Government (2009)).

Type of agricultural production	Total economic output
Sheep production	£165 million (16.5 %)
Cattle production	£ 550 million (55 %)
(excluding milk and milk related product sales)	
Total agricultural production	£1 billion

The dependency of Scottish agriculture on livestock farming is an outcome of limited land capability in the country. 69% of the land area of Scotland is suitable only for improved grassland or rough grazing according to land capability classifications (JHI, 2015; 2016), and 85% of Scottish agricultural land (and 64% of agricultural holdings) has Less Favoured Area (LFA) status (Scottish Government, 2009a). Most cattle production, 90% of dairy and 80% of beef, takes place in LFAs as demonstrated in Figure 5.1, which shows the strong overlap between LFAs and livestock farms.

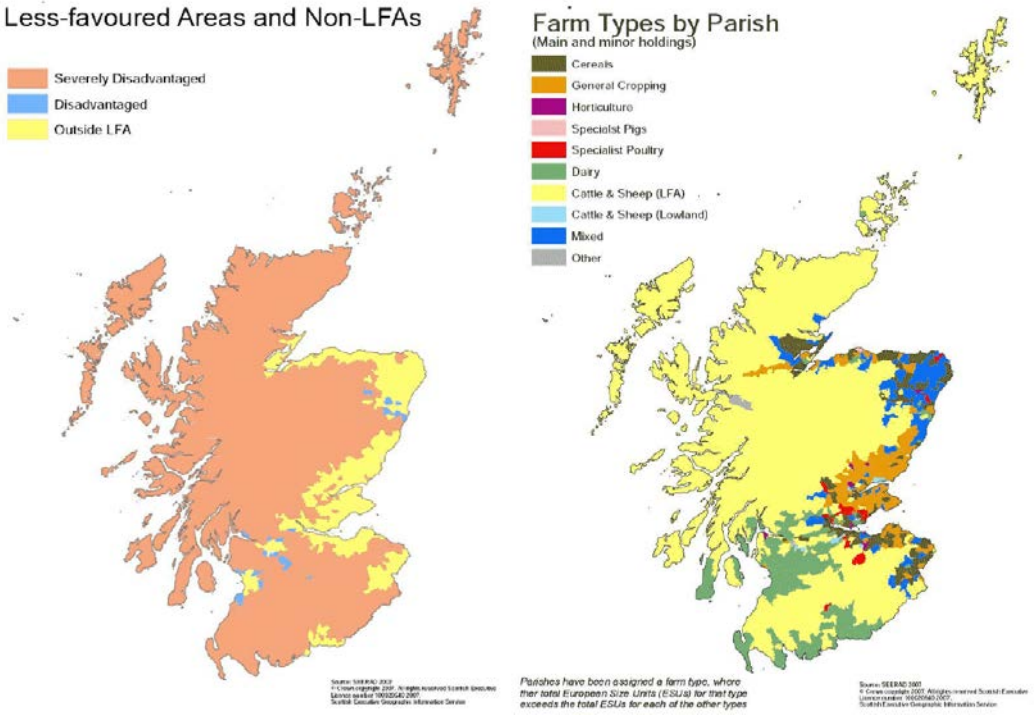


Figure 5.1. Distribution of LFAs and farm types across Scotland (Scottish Government 2016a).

According to Scottish Government estimates, 53% of 13,000 farm businesses are categorised as being dependent on cattle and sheep activity for more than two-thirds of their income and a further 10 % are dependent on dairy farming. Mixed farms are those with no single crop or livestock which accounts for two-thirds of the income. Dairy farming mostly takes place on specialised dairy farms, though almost 40% of the nation’s beef cattle is concentrated in Aberdeenshire where most farms are mixed. Mixed farms may also have herds as large as specialised livestock farms. While animals, especially dairy cattle, are mostly concentrated in the Southwest (Figure 5.2) across less than 15% of the farms, most farms in the Northern regions have herds smaller than 50 animals which are predominantly for beef (Scottish Government, 2016e, 2016h).

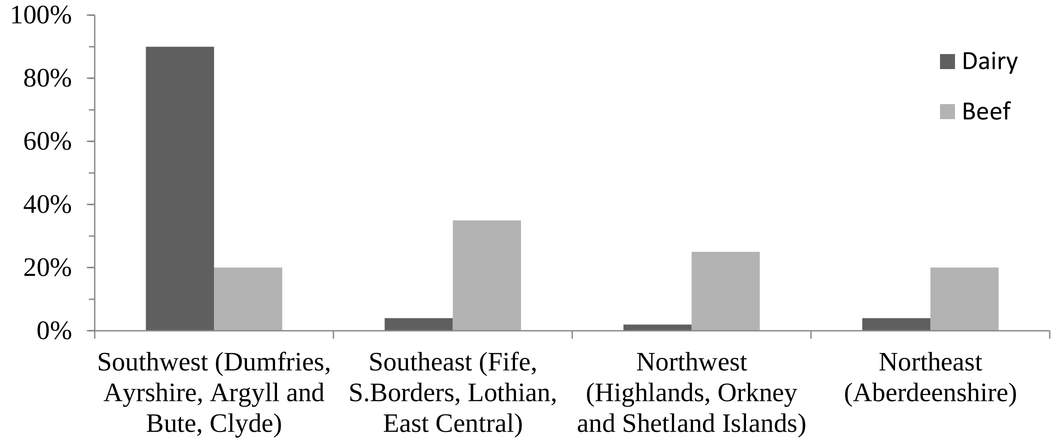


Figure 5.2. Distribution of livestock across the regions of Scotland (adapted from Scottish Government (2016)).

Despite a significant decrease in the number of dairy cows and the disappearance of half of Scotland's dairy holdings since the early 1980s as a result of a substantial rise in the cost of fuel, fertilisers and feed, dairy farming remains important in Scottish agricultural production (SNH, 2016). In comparison to dairy cattle, beef cattle are more evenly distributed across the country (Figure 5.2). However, altitude determines the farming type in terms of lowland, upland and hill livestock systems. Lowland livestock and dairy farms are mostly found in Ayrshire, Dumfries and Galloway, the Borders, Orkney, Caithness and parts of Tayside and Grampian (Scottish Government, 2016e). Unlike dairy farms that have fixed contracts with milk product companies and supermarkets, beef is mostly sold deadweight in spot markets. This introduces greater fluctuations in meat prices compared to milk, as most beef farmers lack significant volumes of stock to trade and they do not have the power to negotiate prices in spot markets (Scottish Government, 2014a; SRUC, 2016). This makes it harder to produce robust value estimates related to livestock farming over time. Without subsidies, livestock production in the Scottish uplands and hills would not be profitable, leading to a severe decline in livestock farming and unavoidable spillover effects on the rural economy, society and environment in these areas (QMS, 2016). To prevent such outcomes, support programmes such as the LFA Support Scheme and the Single Farm Payment (SFP) are used.

5.3 Water use characteristics on livestock farms

Water use related to livestock drinking, slurry flush systems, animal and housing hygiene, feedstock irrigation and domestic use takes place on both dairy and beef cattle farms

(Beckett and Oltjen, 1993). However, water use is more intensive on dairy farms as a result of additional activities related to milk production such as plate cooler water, yard and parlour cleaning and wash down, cleaning of bulk tanks and parlour plant that requires significant volumes of water (AHDB, 2007). In addition, there are considerably higher drinking water requirements for lactating cows as water is the largest constituent of milk at 78% (University of llinois, 2017). Figure 5.3 summarises the different demands for water use on livestock farms.

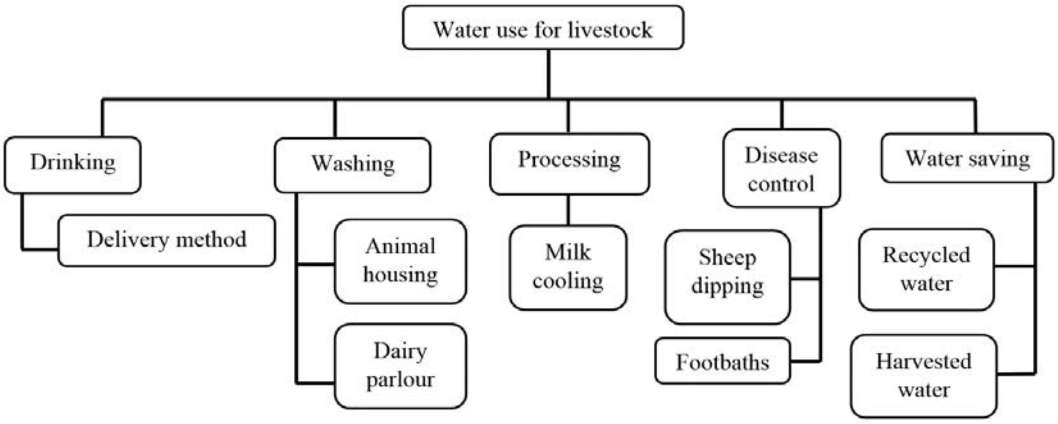


Figure 5.3. Possible water use pathways on livestock farms (Warwick University, 2008).

Using water use characteristics, water use data were estimated as described in Section 5.3.1 on dairy farms and in Section 5.3.2 on beef farms to input to later calculations.

Livestock animals go through specific life sequences, detailed in Figure 5.4 for dairy and beef cattle. Herds comprise animals at different life stages at ratios depending on the farm type, size and management approach and there are different methods of managing cattle systems (FAO, 2017). Herd compositions are dynamic depending on culling and replacement rates of the herd. Here a mixed approach of both buying young cattle for short keep at auctions and keeping calves for background breeding is assumed to provide a representative herd for beef farms. For dairy farms a milking herd without growth (culling rate=replacement rate of 30% annually) is assumed. Thus, the number of weaned heifers kept on the farm for background breeding is around 1/3 of the milking herd (sum of lactating and dry cows).

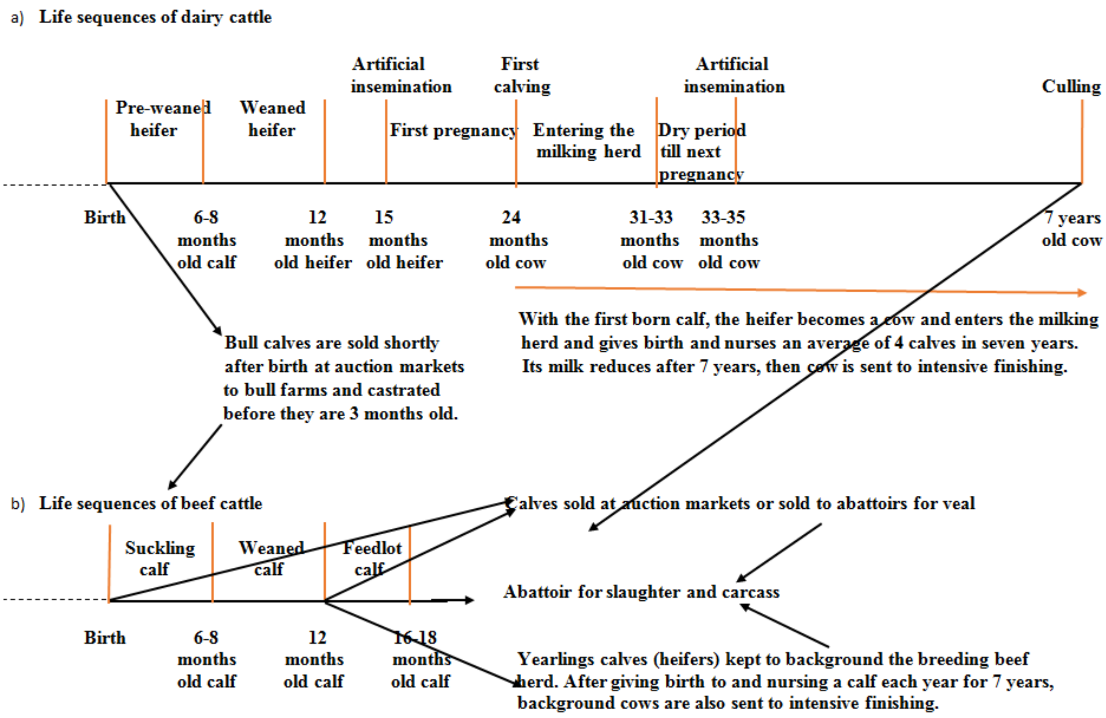


Figure 5.4. Sequences in the life cycle of dairy and beef cattle

Average herd sizes tend to be smaller (97 animals for dairy and 47 for beef in 2014) in Scotland (AHDB, 2016) compared to other countries. In this research herds of 100 animals were considered for both dairy and beef farms because dairy farm profitability data from SRUC Farm Management Handbook used in the estimations is based on a 100 animal herd (SAC Consulting, 2016). Herd compositions are used in both calculations are summarised in Table 5.2.

Table 5.2. Dairy and beef herd compositions used in the profitability estimations.

Type of livestock farm	Type of animal	Number of animals
Dairy farms	Lactating cows	50
	Dry cows	20
	Weaned heifers	20
	Suckling (pre-weaned) heifers	10
	Total herd size	100
Beef farms	Lactating cows	13
	Short keep calves and cows*	60
	Weaned (background) heifers	14
	Suckling heifers	13
	Total herd size	100

*Half of the short keep animals are assumed to be weaned calves and bulls and the other half cows to be intensely finished.

5.3.1 Water use on dairy farms

Using data from the Milk Development Council (2007), the percentages of water use are estimated for different stages of dairy production without any reuse or recycling. Depending on farm type, size and technology, the percentages may vary and water use efficiency in some processes related to washing of parlours and tanks tends to increase significantly on farms that adopt new technology. Thus, farms are categorised in two groups based on adoption of water efficient technologies, assuming the best water use efficiency (category 1), and the worst water use efficiency (category 2) scenarios. The share of each on-farm activity in the overall water use under both scenarios is listed in Table 5.3. Irrigation of feedstock is not considered as it is not common in Scotland. An average of category 1 and category 2 farms is assumed as the representative water use pattern of a standard herd of 100 animals because the exact frequency of category 1 and category 2 farms among the Scottish farms is not known. These average water use figures on dairy farms used in the later estimations are summarised as daily and annual figures per animal in Table 5.4.

Table 5.3. Water use percentages among different activities on dairy farms (figures adapted from MDC (2007)).

Activity on dairy farm	Category 1 dairy farm	Category 2 dairy farm
Livestock drinking	75%	50%
Plate cooler water	15%	25%
Yard and parlour washing down	5%	15%
Plant washing	4%	9%
Other	1 %	1%

Table 5.4. Average daily water use on a dairy farm (AHDB Dairy, 2007; Ward and McKague, 2007) with annual water use per animal shown in parentheses.

Water use on a dairy farm	Average daily and yearly water use (l/animal)*
i) Drinking water requirement by age**	
Lactating suckler cows (over 24 months)	115 (41975)
Dry cows (over 24 months)	40 (14600)
Weaned background heifers (3-24 months)	25 (9125)
Suckling heifers (0-3 months)	9 (3285)
ii) Hygiene	
Cleaning of animals, footbaths, housing, slurry	21 (7665)
iii) Milking process related hygiene	
Plate cooler, collecting yard, plant washing, other	68 (24820)

*may fluctuate through the year with ambient temperature, dry matter content of the feed and the weight of the animal. **derived from annual values obtained for a 100 animal herd.

From these daily figures, the average annual water consumption of a dairy farm with a herd of 100 animals that includes a milking herd of 50 lactating cow will be 7,470 m^3.

5.3.2 Water use on beef cattle farms

The global average water footprint of beef sold in consumer markets is 15,400 m^3/tonne (Mekonnen and Hoekstra, 2012) and has received a lot of media attention as being greater than any other food product. In this analysis, water use per animal will be limited to the lifecycle of beef cattle up to the farm gate and is based on the average drinking water (without consideration of feedstock content) and basic hygiene requirements as listed in Table 5.5 below. Water use for feedstock irrigation and food processing activities related to slaughter houses are not considered.

Table 5.5. Daily average water use on a beef cattle farm (Ward and McKague, 2007) with annual water use per animal shown in parentheses.

Water use type	Average daily and yearly water use (l/animal)*
i) Drinking water requirement by age*	
Lactating suckler cows (over 24 months)	70 (25,550)
Dry cows (over 24 months)	40 (14,600)
Weaned background heifers (3-24 months)	25 (9125)
Growing/fattening short-keep calves (3-24 months)	41 (14965)
Suckling heifers (0-3 months)	9 (3285)
ii) Hygiene**	
Cleaning of animals, footbaths, housing, slurry	21 (7,665)

*may fluctuate through the year with ambient temperature, dry matter content of the feed and the weight of the animal. The values averaged over a year is taken from Ward and McKague (2007).
**derived from annual values obtained for a 100 animal herd.

From these daily figures, the average annual water consumption on a beef farm with a herd of 100 animals will be 2,405 m^3.

Water use per animal information for both dairy and livestock systems was taken from a factsheet (Ward and McKague, 2007) using values previously constructed for Northeast US (Adams, 1995; McFarland, 1998). The Northeast region as defined by the US Census Bureau comprises of nine states: New York, New Jersey, Vermont, Massachusetts, Rhode Island, Maine, Connecticut, New Hampshire, and Pennsylvania (US Census Bureau, 2013). These states have similar annual precipitation figures and the arithmetic average is 1150 mm/year (Daly et al., 2003) compared to 1441 mm/year in Scotland (UK Met Office, 2017a). Also the average summer temperatures in the region are higher than in Scotland (NOAA National Climatic Data Center, 2017), thus drinking water demand of cattle is expected to change more over the seasons. Therefore, rather than sessional ones, average annual water use values were used for both dairy and beef cattle. Hygiene related water demand is taken from AHDB Dairy (2011) and also applied to beef system excluding the milking processes related water demand.

5.4 Methodology and data

A lack of markets requires adaptation of deductive methods to estimate the value use of water on livestock farms (Massarutto, 2011). Due to its less complicated data requirements, netback methodology is chosen as the methodology over other deductive methods, such as hedonic pricing or optimisation models, to value water use in livestock production in Scotland. The methodology, which is a variation of the residual method, derives a shadow value of water as an input to production because the market's mechanisms for water are not efficient enough to dictate prices that reflect the full value of its use. Previous applications of this method to agricultural water uses have considered irrigation use (Ahearn and Vasavada, 1992; Bate and Dubourg, 1997; Lindgren, 1999; Brunnstrom and Stroemberg, 2000; Macgregor et al., 2000; Nilsson et al., 2003). (Ahearn and Vasavada, 1992; Bate and Dubourg, 1997; Lindgren, 1999; Brunnstrom and Stroemberg, 2000)

The netback for livestock production reflects the farmer's maximum ability to pay for water, which might be slightly different to the willingness to pay due to the shadow price of some production factors not considered or underestimated in the cost and profit accounts of farms (Nilsson et al., 2003).

Equation 5.1 and its descriptive notation below explain the netback methodology:

$$\mathrm{N_w} = (\mathrm{P_f} \times \mathrm{Q_f}) - \mathrm{C}_{nonW} - \mathrm{S} \qquad (5.1)$$

where:

N_w: netback of water use, farmer's net profit in £ after costs of production factors excluding water related ones
P_f: the market-clearing price of the farmer's unit output at the farm gate, £ per kg of beef or litres of milk
$Q_{f:}$: quantity of the farmer's milk or beef output sold to the market (kg or litre)
$C_{non\text{-}w}$: all costs involved in maintaining the farm and production except costs related to water in £
S: total net subsidies that farm receives for support in £

Incremental return on a dairy farm is estimated by the litres of milk produced multiplied by the average wholesale market price of milk at the farm gate. While on a beef farm, the kilogrammes of meat carcass sold annually is multiplied by the average wholesale market price of deadweight meat at the farm gate to estimate the revenue. The required data is assembled for 100 animal herd operating under efficient and inefficient water use scenarios as the representative farm model (Gittinger, 1982). This is to match with production scenarios based on a partial farm budget of variable non-water costs and estimated fixed costs (overheads) and to construct £/m^3 value estimates for water use on livestock farms. Once non-water overheads and variable costs are deducted, subsidies also have to be discounted from the net profits in order to deduce the actual net-profit of a farm's output before any external support, as indicated by Equation 5.2. What remains is the net-profit from the water use and is assumed to be the maximum ability of the farmers to pay for water.

The netback includes the opportunity cost of allocating the water to another user (Bate and Dubourg, 1997) as indicated in Equation 5.2. In an economically optimal case, the netback of water allocated through a market mechanism should be at least equal to its opportunity cost.

$$\mathrm{N_w} - \mathrm{OC_w} = 0 \qquad (5.2)$$

where:

OC_w: opportunity cost of allocating water to the particular activity of dairy or beef cattle production

Under absent or imperfect market conditions, the difference between N_{w-} and OC_w of a particular water use can be used to guide when reallocation decisions are required. In this work, maximum ability to pay figures are compared to the current cost of water under various supply options to identify the disparity between the value and price.

5.4.1. Estimation of profit

There are a number of non-water costs in cattle farm accounts such as labour, fertiliser, pesticide, management and repair costs, taxes, rent for land, buildings and machinery where applicable, electricity and gas, concentrates, forage and bulk feed and miscellaneous expenses related to livestock keeping (SAC Consulting, 2016). Profit margins are variable depending upon internal factors such as the farm system (e.g. low input vs. high output or hill, upland and lowland) and the overall management, as well as external factors such as commodity markets, weather conditions and policy. Years with bad weather lead to shortages of forage and make the technical performance and the farmer's management expertise increasingly critical for profitability in terms of rearing more calves per cow and to heavier weights. Using less purchased feed through better grassland management becomes more important, differentiating average and top performing farms in profitability by achieving higher output with similar fixed costs (Young and Loomis, 2014). However these internal factors could not be taken into account in this study since the statistics included in the SRUC Farm Management Handbook and collected for the Agricultural Census by the Scottish Government do not provide separate records for average and high performing farms as farm performance can vary year to year depending on external factors. Therefore average total costs of production were estimated for each livestock farming system as detailed in the methodology and data sections below.

The Scottish Government provides various subsidies to support farmers against price volatility and provide stability. This study focuses on the direct support of the Scottish Suckler Beef Support (SSBS). Only SSBS that applies to suckler beef production across Scotland is accounted for because subsidies that target specific regions, farming systems or special circumstances, such as new-entry, could not be considered for the standard herd of unspecified location used in this study.

5.4.1.1 Estimation of the profitability of a standard dairy farm

The main dairy farm costs are feed (concentrates) and energy related, such as electricity, gas and fuel for vehicles. Annual average milk yield per cow consistent with yearly profit data is used rather than the lactation cycle that would give monthly milk output. From the annual average yield, milk consumed on the farm is discounted to obtain the final amount sold to the market. Profit made at an average market price is 28 pence per litre for Scotland (SAC Consulting, 2016). This exercise assumes that all livestock farmers receive sufficient technical advice through the Rural Payments Directorate and other relevant organisations such as SAC Consulting that provide support to farming in Scotland. This assumption serves to offset the possible distinctions among farms due to hard-to-price attributes, such as managerial skills, capital requirements, forage prices sensitive to local availability and risk associated with low yield and fast perishability of the produce, all of which would complicate the application of netback methodology (Young and Loomis, 2014). Table 5.6 summarises the data acquired from the SRUC Farm Management Handbook for costs related to production on a dairy farm (SAC Consulting, 2016).

Table 5.6. Estimation of non-water variable costs for dairy farms adapted from (SAC Consulting, 2016).

	Low input-low output dairy farms (5,000-7,000 litres per cow per year)			**Medium to high output dairy farms (8,500-10,000 litres per cow per year)**		
	Profitability (pence per litre, ppl)	**%**	**£/year for 50 lactating cows (at 5,000 litres)**	**Profitability (ppl)**	**%**	**£/year for 50 lactating cows (at 8,500 litres)**
Milk price	28.00	100	140,000	28.00	100	238,000
Concentrates	-7.00	-25	35,000	-8.40	-30	71,400
Forage and bulk feed	-2.10	-7.5	10,500	-2.38	-8.5	20,230
Sundry livestock	-2.10	-7.5	10,500	-2.38	-8.5	20,230
Gross margin	15.80	60	84,00	13.16	53	126,140
Labour	-3.36	-12	16,800	-3.36	-12	28,560
Power (fuel)	-4.20	-15	21,000	-5.04	-18	42,840
General overheads	-1.40	-5	7,000	-1.4	-5	11,900
Gross profit before rent	7.84	28	39,200	5.04	18	42,840
Rent, finance and quota	-1.96	-7	9,800	-1.96	-7	16,660
Net profit*	**5.88**	**21**	**14,700**	**3.08**	**11**	**13,090**

*The business cost structure may vary significantly between individual farms and farming types.

A dairy farm with a herd of 100 cows, of which 50 are lactating, would make an average annual profit between £13,090 and £14,700, depending on the type of herd (low input or high input). The mean profit of the figures in Table 5.6 is £13,895.

5.4.1.2 Estimation of the profitability of a standard beef farm

Costing figures as well as market prices for beef were taken from the SRUC Farm Management Handbook (2016) to estimate the profitability on beef farms. Missing fixed costs are transferred from the lower bound of dairy farms, which tend to be more costly to maintain. Unlike dairy systems where profits can be directly linked with the amount of milk sold, beef systems are complicated as there are parts of the system, such as calves, that add to the costs but not to the immediate profits. Therefore, the profitability is calculated on an average herd and values across spring and autumn seasons as well as the aggregation of hill, upland and lowland systems. It is also assumed that output per hectare and expenses are similar across the farm categories due to data limitations and that all prime beef cattle are marketed deadweight in spot markets. It should be noted that, in practice, 80% of beef cattle are sold as deadweight and the remaining 20% is marketed through live auctions. Table 5.7 summarises estimations of different components of beef herd profitability.

Table 5.7. Gross margin for various animals on a beef cattle farm based on per animal profit figures from SRUC Farm Management Handbook (2016).

Cattle type	**Steer (£)**	**Heifer (£)**	**Average (£)**
Spring calving cows producing 18-20 month finished cattle	314.00	256.00	285.00
Overwintering calves	66.00	60.00	63.00
Finishing spring born suckled calves at 12 months	44.00	157.00	100.50
Finishing autumn born suckled calves at 18 months	11.00	-34.00	-22.50
Finishing spring born suckled calves at 18-20 months	68.00	89.00	78.50
Beef cattle summer finishing	70.00	10.00	40.00
Intensive dairy bred bulls (Holstein and Continental)	-45.00	93.00	24.00
Calves up to 12 months average	55.00	108.50	50.25
Calves up to 20 months average	123.67	103.67	113.67
Cows average	12.50	51.50	32.00
Farming type for suckler cows	**Hill**	**Upland**	**Lowland**
Suckler cows	230.00	287.00	133.00
Scottish suckler beef subsidy scheme	78.00	80.00	80.00
Non-LFA region net profit from liveweight sale	-	-	53.00
LFA region net profit from live weight sale	152.00	207.00	-
LFA region average net profit from live weight sale	179.50	-	-

Using the average figures above, keeping a beef herd of 100 animals on a farm composed of 27 background livestock, 13 suckling cows and 60 short-keep cattle and calves provides an annual profit of £4,212 based on their sale.

5.5 Water pricing scenarios on livestock farms

Traditionally livestock have been allowed to drink from any available water source around the pasture but this practice has come to an end with the Scottish Controlled Activity Regulations (CAR) that prevent animals going within 5 metres of any water source (Scottish Parliament, 2011). With this additional water demand, the volume required and thus cost for water provision has increased for cattle farms.

Water supply options on a cattle farm are mains water supply and different forms of private water supply (groundwater abstraction, surface water abstraction and rainwater harvesting). The cost per m^3 of acquiring water differs between options and most farms diversify, rarely relying on one supply option. The most expensive option is mains water, which in the UK has an average charge of £0.99/m^3. However, 2014/15 statistics in England and Wales show that an average cattle farm sourced two thirds of its water from the mains supply, 18% from bore holes and abstracted 12% from rivers/streams/springs for immediate use (Defra, 2016). While mains supply is usually preferred by smaller, lowland grazing and dairy farms, remote hill and upland beef farms that are situated in LFAs and larger farms instead opt for private abstraction. Although water from private supplies is less expensive than mains, it is not free and may incur costs such as purchasing and maintaining pumping equipment.

The range of water costs will be even wider on dairy farms compared to beef farms because of higher water demand on dairy farms related to the milking process and larger average herd sizes. Water costs among dairy units will vary considerably with local conditions, such as the availability of boreholes in the region, and those related to the size of the farming business, such as the average volume demanded per day. For a realistic picture of the situation, several scenarios are constructed here based on average water use requirements for the same structure of dairy and beef herds used in the above estimations.

5.5.1 Mains water supply option

Mains water supply used on any premises in the UK is regulated by the Water Supply (Water Fittings) Regulations (1999) or Byelaws (2014) in Scotland. These regulations prevent the misuse, waste, incorrect measurement of consumption and, most importantly, contamination of mains from non-mains sources, including harvested rainwater (Scottish Parliament, 2014).

The cost of mains water supply for business water users is assumed to be an average of £0.8042/m^3 up to 100,000 m^3 (SWBS, 2013; United Utilities Scotland, 2016). Annual water demand is unlikely to exceed that volume for dairy and beef cattle under both category 1 (efficient) or category 2 (inefficient) water use scenarios described above.

5.5.2 Abstraction supply option

From 2005 onwards, abstractions require licensing either by the Environment Agency in England, Natural Resources Wales in Wales or SEPA in Scotland. Operating costs associated with a private water supply originate from licence application and annual subsistence charges depending on the m^3/day abstraction amount.

Any abstraction will require a licence that ensures that there is no negative impact to groundwater and surface water bodies and must be registered with SEPA under the Water Environment (Controlled Activities) (Scotland) Regulations (CAR) (2011). The updated CAR regime follows a tiered approach with the thresholds for abstraction volumes indicated in Table 5.8. The initial and annual cost associated with each tier is different.

Table 5.8. Abstraction licensing regime in Scotland according to CAR (adapted from SEPA 2014).

Daily abstraction volume	Licensing requirement
<10 m^3	General binding rules
10-50 m^3	Registration
50-2000 m^3	Registration and simple licence
> 2000 m^3	Registration and complex licence
All new impoundments	Registration and complex licence

It is assumed that for an average dairy or beef herd of 100 cattle, the total daily water requirement would be met with a licence registration with SEPA, enabling up to 50 m^3/day at a one off cost of registration without annual subsistence charges. Even with abstractions from as many as 25 points, providing up to 2000 m^3/day, which is unlikely for an average livestock farm, the annual subsistence charges do not currently exceed £625.26 charged for basic non-irrigational agricultural water abstraction license detailed in Environmental Regulation (Scotland) Charging Scheme (SEPA, 2016a).

Although complex licences might be required by farms with larger herds, it is not considered as the norm for a farm with a herd of approximately 100 cattle. Licences are granted by the Environment Agency in England and Natural Resources Wales in Wales for a period of between 6 and 18 years initially and renewed for another 12 years (UK Government, 2014). In this study, it is assumed that the framework in Scotland is similar and the cost of licence application and registration is amortised over the first 12 years. The capital costs of infrastructure required prior to licence application, fees and operational costs are calculated according to the costing items listed in MDC audit (2007; 2011) for effective water use on farms. The working life of the pumping system is assumed to be 25 years and straight-line depreciation is applied. The annual costs related to abstraction are listed in Table 5.9.

Table 5.9. Overall annual cost of water abstraction for a Scottish livestock farm with a herd of 100 animals. Lists of costs taken from AHDB Dairy (2011) and adjusted to 2016 prices.

Cost items related to abstraction	**Cost (£)**
i) Capital and one off costs	
Geological report for initial investigation	580
Test bore	580
Main bore	5800
Borehole pump	580
Pump shed	174
Pipe line from supply to farm (@ 1 £/m)	116
Electrics, tanks, pumps, pressure vessels, pipe work, filters, etc.	1232
SEPA abstraction licence application fee	82
SEPA (simple) abstraction licence registration fee	612
Cost of local advert by the applicant*	500
Total (one-off) infrastructure costs	**10,256**
Annual depreciation cost (over 25 years)	**410.25**
ii) Annual operational costs	Small scale
SEPA annual subsistence costs	625.26
Pump service/maintenance and cost	0
Annual labour costs for maintenance	58
Total annual operational costs	**683.26**
iii) Annual electricity costs	
Pump size kW	3 kW
Hourly cost of electricity (£/kWhr)**	0.42
Pump run time (hr/yr)**	1800
Annual electricity cost of abstraction (£)	**756.54**
Total cost of abstraction (£)	**2,055.55 (1,439.8 operation cost)**
Estimate of annual abstracted water volume (m^3)	7470 for dairy and 2405 for beef herd
Average annual cost of abstraction (£/m^3)	**0.24 for dairy and 0.85 for beef herd**

*Where the disposal could have an impact on parts of the water environment associated with others, e.g. a stretch of fishing waters, you may be required to advertise the application, at your cost, before an authorisation decision is made (SEPA, 2011b)**£0.20 is the daily baseline charge for the electricity adding up to an annual £7.30 additional and fixed baseline cost independent of run time.**14.1p/kWh (Standard tariff) of Scottish and Southern Electricity average tariff.*** Figure is estimated based on 5 hours daily operation and 360 operating days a year.

5.5.3 Rain collection (harvesting) option

Rainwater harvesting is a sustainable alternative for reducing dependence on mains water and abstraction. Although it involves investment for set up due to the alterations in infrastructure required (gutters and down pipes, and the electrics, storage tanks, pumps, pressure vessels, pipework, filters), the operation and maintenance costs are minimal (AHDB Dairy, 2011). Based on the average annual rainfall for the period 1910-2016 for the UK and Scotland of 1097 mm and 1441 mm, respectively (UK Met Office 2017a, UK Met Office 2017b), with 80% storage efficiency, it would be possible to collect annually 878 m^3 in the UK (1153 m^3 in Scotland) from when roof area of 9-10 m^2 $animal^{-1}$ is available for water collection (Thompson et al., 2007). With a 100 animal herd, the average Scottish figure would be around 12-36% of the on-farm water demand depending on whether it is a beef or dairy farm. The working life of a rainwater harvest system is assumed to be 25 years and the costs are amortised accordingly over the years. The cost ranges between £0.2 and £0.5/m^3 according to figures from AHDB Dairy (2007) adapted to 2016. Within this range the cost on a specific farm would depend on the size of capital investment and amount of water collected. The rate of return on capital is dependent on the usage intensity and rainfall availability, however rainwater harvesting is only a complementary method for water use efficiency and cost reduction on site. Even in a country like Scotland with a comparatively wet climate and consistent precipitation, the water supply of a farm business cannot fully rely on rainwater harvesting.

5.6 Results

In this study, how much value is created per each cubic metre of water used in the dairy and beef cattle industry in Scotland in the current circumstances has been analysed. The findings of the analyses described above are summarised in Table 5.10

Table 5.10. Annual water use, profit figures, estimate for value of the water and the cost of the water based on 100 animal dairy and beef herds as described earlier.

i)	**Annual water use**	**m^3**
Dairy		7,470
Beef		2,405
ii)	**Annual profit made**	**£**
Dairy		13,845
Beef		4,212
iii)	**Economic value of water use**	**£/m^3**
Dairy		1.9
Beef		1.8
iv)	**Cost of water use**	**£/m^3**
Mains		0.80
Abstraction		0.24 (dairy), 0.85 (beef)
Rain harvesting		0.20-0.50

According to the results reported in Table 5.10 the dairy industry has a higher valuation of water use compared to beef due to the higher profits made on dairy herds compared to beef herds. Moreover, the results show that the actual value of water to its users, both in dairy and beef production, is much higher than the cost of any supply option available, which indicates a fundamental mismatch between the cost and value of water in the livestock industry in Scotland.

Fluctuations in profitability of farm businesses are influenced by several external factors such as market demand and supply conditions, prices for both inputs and outputs, interest rates for long-term investments and overall productivity changes due to climate and technological change (FAO, 1998). These, combined with uncertainties of up to 15% in drinking water requirements, general intensity and productivity in cattle systems (Chatterton et al., 2010), introduce a certain degree of sensitivity to the estimated results. Moreover, other factors of production not considered in the reference sources of information, such as financial stability and the scale of the farm or the technical expertise and experience of the farmer in management, may lead to an over-estimation of the £/m^3 values of on-farm water use.

While the results do not have high precision, the estimated values for both dairy and beef farms are still of considerable importance, being the first publicly available figures for the livestock industry. The results here indicate considerable difference between the economic value of water use and the prices of different water supply options available to farms. Even the most expensive supply option, mains water, at £0.80/m^3 in Scotland, is much cheaper than the estimated value of £1.9/m^3 and £1.8/m^3 on dairy and beef farms, respectively.

The reason why the m^3 cost of abstraction and harvesting options are much higher for beef farms than dairy farms in the above estimations is the fact that for the same initial capital investment, more water is used on dairy farms, making the unit consumption cost cheaper. While rain harvesting is lower cost than mains water supply, the cost is quite similar to abstraction for small volumes. Thus, making an additional investment for rain harvesting when there is already abstraction infrastructure in place would be unnecessary for smaller enterprises. From the estimates above, it also appears that rainwater harvesting alone would not meet the full demand for the standard herd and needs to be supplemented with other supply options.

5.7 Conclusions

Livestock is a water-intensive industry that is projected to grow due to increasing demand for meat and dairy products globally (Thornton, 2016). However, the economics of water use in the livestock industry have not been well-studied, despite its relevance to current water policy and the projection of increased competition for water users in the future as a result of climate change.

This study estimated the economic value created as a result of water use on cattle farms in Scotland to provide a unit value (£/m^3) proxy comparable to water used by competing water industries, in order to inform policy related to water allocation decisions. The current price scenarios estimated for water supply highlight the gap between the price and value of water use on livestock farms. There are several important deductions from this analysis.

Dependency of the livestock industry on water supply is apparent. The results of the netback analysis show that the economic value of water use in livestock is as high as some industrial water uses and much higher than the value transferred for agricultural irrigation in the Chapter 2 on current allocation and valuation of water in Scotland. The charges associated with the provision of mains water and licensing of private supplies partly reveal the cost of managing water from the supply side in compliance with the Water Framework Directive (European Commission, 2000). However, it is still far from reflecting the full value of volumetric water use in line with economic principles. The apparent disparity between the cost and price of water in the livestock industry in Scotland indicates that farmers could pay more for their water use. A potential increase in water prices would also increase the water productivity on farms. Reforming distribution of abstraction licences in a way to match the value would incentivise more sustainable water use options, such as rain harvesting and the upgrading of recycling and reuse technologies on farms.

The Scottish farming industry has become increasingly reliant on subsidies (Scottish Government, 2016g). Those directly relevant to beef and dairy cattle production are Less Favoured Area Support, New Entrants, Small Farms Grant Scheme and Scottish Suckler Beef Support Schemes (Rural Payments and Services, 2017). Scottish farmers receive 85% of Less Favoured Area payments and 18% of the UK's overall Common Agricultural Policy (CAP) funding under the EU system (Stewart and Misselbrook, 2017). The Scottish livestock industry and many other industries that rely on its produce face considerable uncertainty regarding subsidies post-Brexit. However, if managed well, the situation might potentially be turned into an opportunity to make farm businesses, which are not profitable without direct subsidies, competitive again (Cox et al., 2017; Skerratt et al., 2016) and to re-consider environmental policy including management and allocation of water resources.

The results here are preliminary as differences at regional and individual farm level are not considered. A sensitivity analysis of milk and especially beef prices could complement these results to reflect the range of uncertainties that might affect current farm profitability resulting from price fluctuations in agricultural markets. Further research in quantifying uncertainties linked with climate change projections and more focus on region specific assumptions, in terms of subsidies and other local factors such as precipitation, will yield results that are more robust.

Chapter 6 Application of market based instruments to diffuse pollution control in Scotland

The candidate collected the data, prepared the code and performed analyses for allocation and valuation of different water uses and wrote the paper. Dr Ermolieva and Dr Rovenskaya supported and guided the scope and design of the project. Dr Ermolieva checked the code and Dr Rovenskaya helped with the construction of the mathematical model. Prof. Heal and Prof. Moran provided feedback on the content and contributed to the editing of the manuscript.

Chapter 6 Application of market based instruments to diffuse pollution control in Scotland

Abstract: Nitrogen is an essential element to enhance plant growth and fertilisers are thus applied to soils for increased agricultural production. However, accessible nitrogen from fertilisers and animal manure can leach into soil, oxidise to nitrate and cause diffuse pollution of water resources. This uncontrolled pollution has a high economic cost to society both in terms of treatment cost, as well as health and environmental damage. Market based tools applied to catchment management can help incentivise a behavioural change in farmers who will not be the direct beneficiaries of the improved water quality downstream and help reduce pollution and its resulting cost for all. In the scope of this paper we design a trading scheme for a case study catchment in Scotland based on a farm income optimisation model that allocates pollution permits. Pollution load allowances estimated to have a market value of £1.4 per kg and for farms that pollute less than their allocated pollution allowances, this can be a significant source of additional income. Thus, farmers are economically incentivised for more efficient nitrogen management on their land so that they can sell their access permits. The total cost of avoided fertiliser use and water treatment is at least £ 500,000 annually in the case study catchment alone. The model results highlight the potential for trading to replace direct subsidies for farm business support.

6.1 Introduction

Although Scotland has abundant and better quality water resources in comparison with other European countries, pressures on water resources are increasing with climate change (SEPA, 2015b, 2016a; SNH, 2015) and more effort has to be concentrated on planning and adaptation in water management. Among all the regulated activities related to water management, diffuse pollution is currently the principal pressure on Scottish freshwater resources and the major obstacle to accomplishing WFD targets (Scottish Government, 2015c).

Diffuse pollution differs from point pollution which is discharged at a definite point or end of a pipe, making the latter more convenient to control and regulate. In contrast, diffuse pollution occurs on extended land surfaces through the leaching of pollutants into surface and ground waters with rainfall, water infiltration and surface run-off. Individual sources of diffuse pollution can be minor and hard to track, yet collectively they may result in significant concentrations of pollutants and considerable impairment of ambient water quality in the receiving water bodies. In Scotland diffuse pollution is mainly caused by rural sources, which are agricultural activities connected directly with land use. Pollution is generated from recent or past uses of various chemicals applied on land, which are then released to water systems with a time lag. In Scotland, 4600 km of rivers, 300 km² of lochs, 80 km² of coastal waters and more than 100 river catchments as well as groundwater aquifers, which feed rivers, are adversely affected by diffuse pollution arising from rural land use.

Due to its environmental and health consequences, monitoring and control of nitrate pollution is already a policy priority in many countries that suffer rural diffuse pollution (European Commission, 1991a; EPA, 1992). Nitrogen is one of the key nutrients applied and is taken up by plants as nitrate to enhance growth. However, in the form of nitrate, it is highly soluble and residual amounts easily leach into the soil and groundwater as well as being carried directly to surface waters in run-off and field drains (Pérez et al., 2003). The nitrate is transported through shallow aquifers and enters springs and streams, thereby increasing the nitrate load in streams and finally in estuaries. Thus, nitrate in groundwater affects the whole water cycle. Nitrate concentrations in rivers and groundwater across Europe are measured, including current levels and historical trends, to identify countries where improvement and/or deterioration in nitrate related water quality take place. Assessment of current concentrations is made against legislative criteria– in the EU primarily a threshold of 50 mg NO_3/l (11.3 mg NO_3-N/l), with a guiding concentration of 25 mg NO_3/l (5.6 mg NO_3-N/l) (European Commission, 1991a, 1998).

Scientific and legislative efforts made in the last 10 years across the EU have not been fully effective to eradicate diffuse pollution from the EU waters partly due to time lags in the appearance of nitrate in the receiving waters (Bouraoui and Grizzetti, 2014) and partly because the pollution control strategy does not take economics into full consideration. Economic tools can provide effective incentives for better land management, resulting in the achievement of improved water and environmental quality at reduced cost for all. In this chapter, it is explored how market based instruments (MBI) can be applied to address nitrate diffuse pollution to complement the current policy and achieve more efficient and effective pollution control. We propose a trading scheme that recommends shifting the capped pollution to the users who produce the most profit as a result of their pollution while compensating the others for polluting less. Thus, the pollution control cost would be minimised and the profit would be maximised at catchment level within the allowed budget.

The chapter is structured as follows. Section 2 provides background on diffuse nitrate pollution and the relevant policy in Scotland. Section 3 elaborates on the role of market based instruments in diffuse pollution control. Section 4 expands on the mathematical description of the trading model and the case study catchment used in the model calibration. Section 5 summarises results from the analysis and Section 6 sets out the conclusions of the analyses.

6.2 Background

The origin of nitrate pollution in rural catchments is usually agricultural activities. Intensive livestock keeping, together with crop cultivation, increases the potential for nitrate pollution in groundwater (Infascelli et al., 2009). It is particularly common and hard to eliminate in groundwater aquifers. The movement of nitrogen pollution in the soil and groundwater is illustrated in Figure 6.1. Of particular note is the transport period, sometimes of several decades (Dunn et al., 2014), between when the nitrogen is applied to the land surface and leached into the groundwater system and when it is detected in sampling boreholes.. This residence time in soil might change with factors, such as distance, soil types, horizons and their hydraulic conductivity, precipitation and frequency of extreme weather events, resulting in a "lag time" between implementation of management actions to reduce nutrient loads and a detectable improvement in surface-water quality (Phillips et al., 1999).

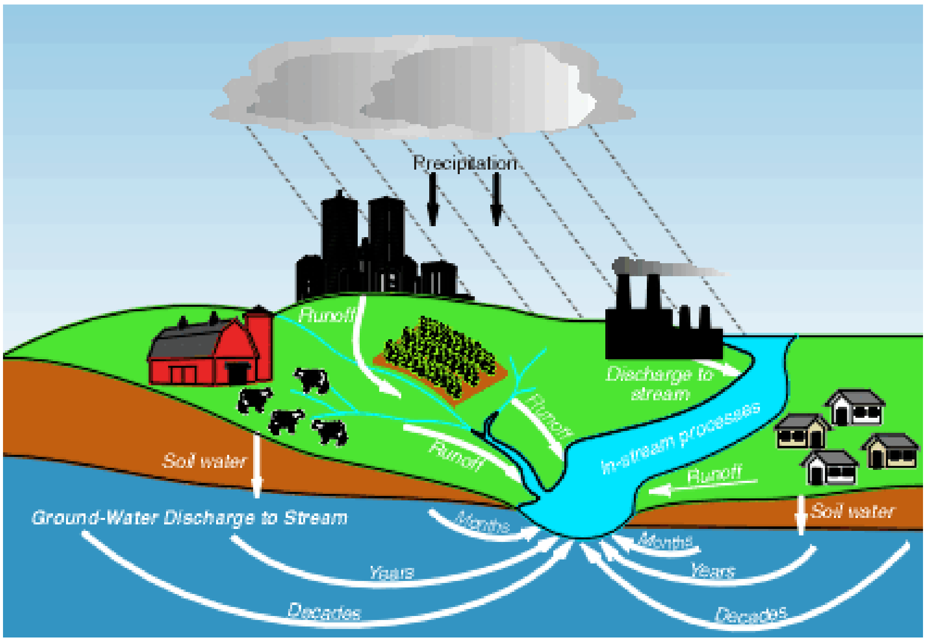


Figure 6.1. Movement of nitrogen in groundwater (Phillips et al., 1999).

Diffuse nitrate pollution has consequences in terms of damage to human health and environment and increased economic losses. High nitrate concentrations in drinking water are known to cause many health conditions including cyanosis in infants, a potentially fatal blood disorder in babies commonly known as "blue baby syndrome" (WHO, 2007), and an increased risk of cancer (van Grinsven, Rabl and de Kok, 2010). High nitrate concentrations also stimulate eutrophication, in which excess phytoplankton growth depletes dissolved oxygen during decomposition in lakes, rivers and coastal waters making them uninhabitable for fish and other species (McClelland and Valiela, 1998; Smolders et al., 2010).

Economic loss due to diffuse pollution is also significant. In the UK alone, the cost to the water industry to reduce high nitrate concentrations in drinking water supplies caused by diffuse pollution has been estimated at €316.5 million in capital and operating expenditures annually for the 2005-2010 period (NITRABAR Project Partners, 2009). These costs are projected to rise as groundwater concentrations and groundwater abstractions for public supply are expected to increase in future (UKGW Forum, 2017). Although translating environmental changes due to increased nitrate concentrations to damage cost functions is complicated (Keeler et al., 2016), an approximate breakdown of the major costs resulting from nitrate pollution is listed in Table 6.1.

Table 6.1. Potential damage costs* of diffuse nitrate pollution to water resources. Most values are from the compilation made by Sobota et al. (2015) and references therein.

Cost** of nitrate pollution	£/kg*	Reference
Health damage		
Increased risk of colon cancer	0.54	(van Grinsven et al., 2010)
Environmental damage		
Reduced recreational use (freshwater)	0.13	Dodds et al. (2009)
Reduced recreational use (coastal)	4.9	Birch et al. (2011)
Eutrophication (freshwater)	0.62	Gren (2011)
Eutrophication (coastal)	0.11	Pretty et al. (2003)
Loss of biodiversity due to eutrophication	0.3	Pretty et al. (2003)
Bad odour and taste	0.14	Kusiima and Powers (2010)
Decline in estuarine and marine habitat	1.31	van Grinsven et al.,(2010)
Decline in fisheries	11.89	Compton et al.(2011)

Economic loss		
Treatment cost	10 to 110	Elliott (2017)
Property value loss	0.21	Dodds et al. (2009)

*Median cost for kg N adjusted to £ in 2017 values, first by converting the figures stated in US dollars or in Euros to 2017 values in the same currency, and then converting these currencies to British pounds in 2017 (Coinnews LLC, 2017; Stat Bureau, 2017; XE, 2017).

Remediation efforts are usually met with a slow response in water quality improvement (Bouraoui and Grizzetti, 2014; Mouratiadou, 2011; RPA Consortium, 2008; SNIFFER 2013; Sohier and Degré, 2010). Additionally, the recovery period including soil and groundwater remediation is costly and might last up to several decades (Phillips et al., 1999), therefore reducing the availability of water to other users and placing a heavy and continuous cost on the water industry. Nevertheless, implementing measures such as land management, manure storage and lower fertiliser application is estimated to be 5 to 10 times cheaper than removing nitrate from already polluted waters (European Commission, 2002). Thus mitigation in the form of pollution control at the source is the most effective strategy to combat nitrate pollution resulting from agriculture.

6.2.1 Policy response to diffuse pollution

The EU Nitrates and Water Framework Directives, specifically and more generically, set in place measures to address costly nitrate pollution. Under the Nitrates Directive (1991) nitrate Vulnerable Zones (NVZs) were initially identified by an EU consultation as coastal, surface and ground waters where nitrate concentrations exceed or are likely to exceed the threshold set in EU legislation (European Commission, 1991a). EU Member States are required to review and report on codes of good farming practice, implementation of additional control measures, results of water monitoring and a summary of relevant aspects of action programmes for designated NVZs to the European Commission every four years. Following the review of the Nitrate Vulnerable Zones (NVZ) Designated Areas consultation, NVZ areas in Scotland from 1 January 2016 comprise Lower Nithsdale; Lothian and Borders; Strathmore and Fife including Finavon; Moray, Aberdeenshire/Banff and Buchan; and the Stranraer Lowlands (Scottish Government, 2016f). These areas, with the exception of Lower Nithsdale and the Stranraer Lowlands, are around the east coast where intense arable agriculture and fertiliser use take place.

The Nitrates Directive requirements are an integral part of the WFD (European Commission, 2000). The WFD also requires River Basin Management Plans (RBMP), which cover an entire river system, including river, lake, groundwater, estuarine and coastal water bodies (Scottish Government, 2015c; Defra, 2016). RBMPs set out how different organisations, stakeholders and communities can work together to improve the water environment in order to achieve the protection, improvement and sustainable use of the water environment across Europe. In Scotland, two RBMPs for the Scotland River Basin District have been published by SEPA, the first covering the period between 2009 and 2015 (Scottish Government, 2009c) with the second current one for the period 2015 to 2027 (Scottish Government, 2015c).

In addition to EU scale measures, WFD has also mandated country specific implications. When the WFD came into force in 2000, it had to be adopted into the national legislation of each member state by the end of 2003. WFD was incorporated into Scots Law through the Water Environment and Water Services (Scotland) Act 2003, later amended in 2006 and 2014 (Scottish Government, 2014c), which acknowledges the general duty of Scottish ministers, environmental agencies and other public bodies to protect water bodies. Stakeholders from across land-use planning, transport, energy, fisheries, enterprise, recreation and tourism, marine and agriculture sectors as well as the water industry were brought together to develop and review water relevant policy and legislation, such as River Basin Management Plans (RBMP).

The Diffuse Pollution Management Advisory Group (DPMAG) is an example of these collaborative partnerships, comprising representatives from various Scottish organisations that have an interest in reducing rural diffuse pollution. The group was established in 2008 to achieve an effective delivery of rural diffuse pollution actions by creating a robust governance, decision-making and coordination framework under the Water Environment (Diffuse Pollution) (Scotland) Regulations (Scottish Parliament, 2008). As a part of the DPMAG Implementation Plan, catchments across Scotland that do not meet the environmental quality standards and require a catchment-wide approach have been classified as first, second and third RBMP cycle priority catchments to reduce diffuse pollution risks. The General Binding Rules (GBRs) within the Water Environment (Diffuse Pollution) (Scotland) Regulations (Scottish Parliament, 2008) were published with the aim of achieving water quality targets by reducing concentrations not only of nitrate but of other pollutants such as suspended sediment, faecal bacteria, in priority catchments.

Catchments across Scotland that do not meet the environmental quality standards have been classified by their urgency as first, second and third RBMP cycle priority catchments for addressing diffuse pollution. The 14 water body catchments in the first (urgent) priority group were selected from over 100 catchments on the basis that they either have protected area status and/or there are already pollution related risks to human health and/or they have more significance in terms of bathing, drinking, conservation and recreation. These catchments will be focused upon in the first six years of the RBMPs between 2015 and 2021, while the second priority group will be tackled in the second six years between 2021 and 2027. Activities conducted in the priority catchments to achieve good status by the end of the cycle in 2027 include better data collection, improved understanding of the dynamics, such as the connection between land use and diffuse pollution and impacts of diffuse pollution, and devising and implementing management measures (DPMAG 2011). Figure 6.2 shows the location of priority catchments for the first and second RBMP cycles.

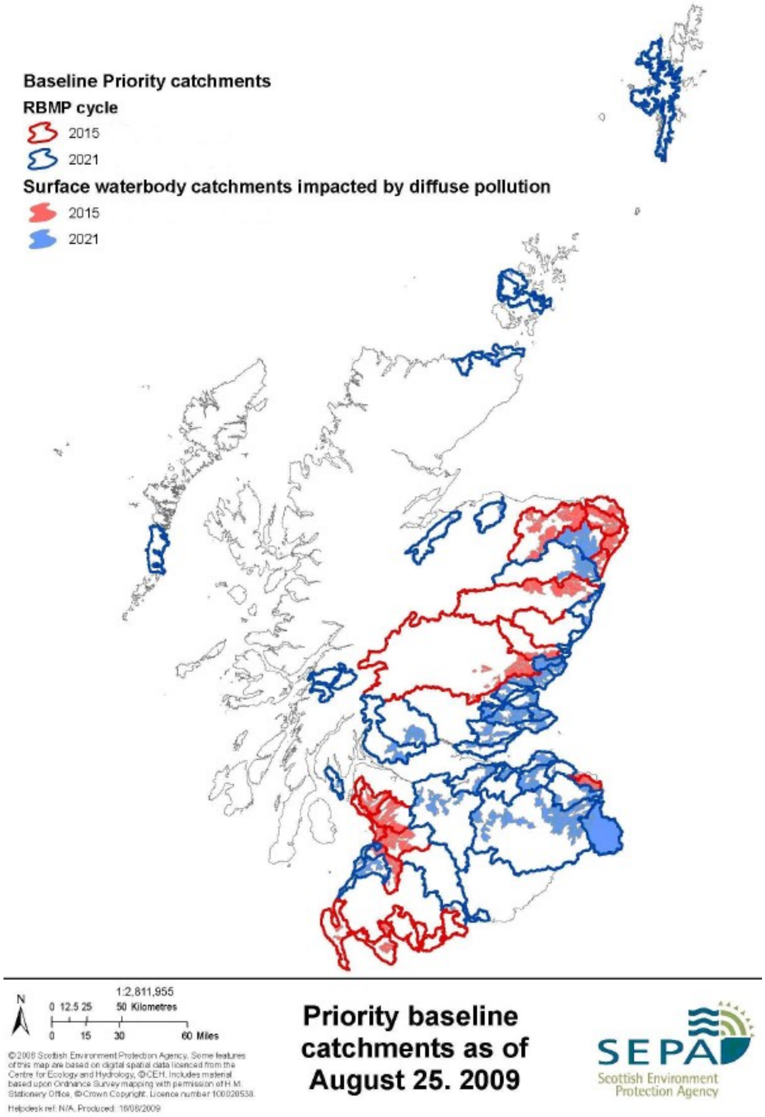


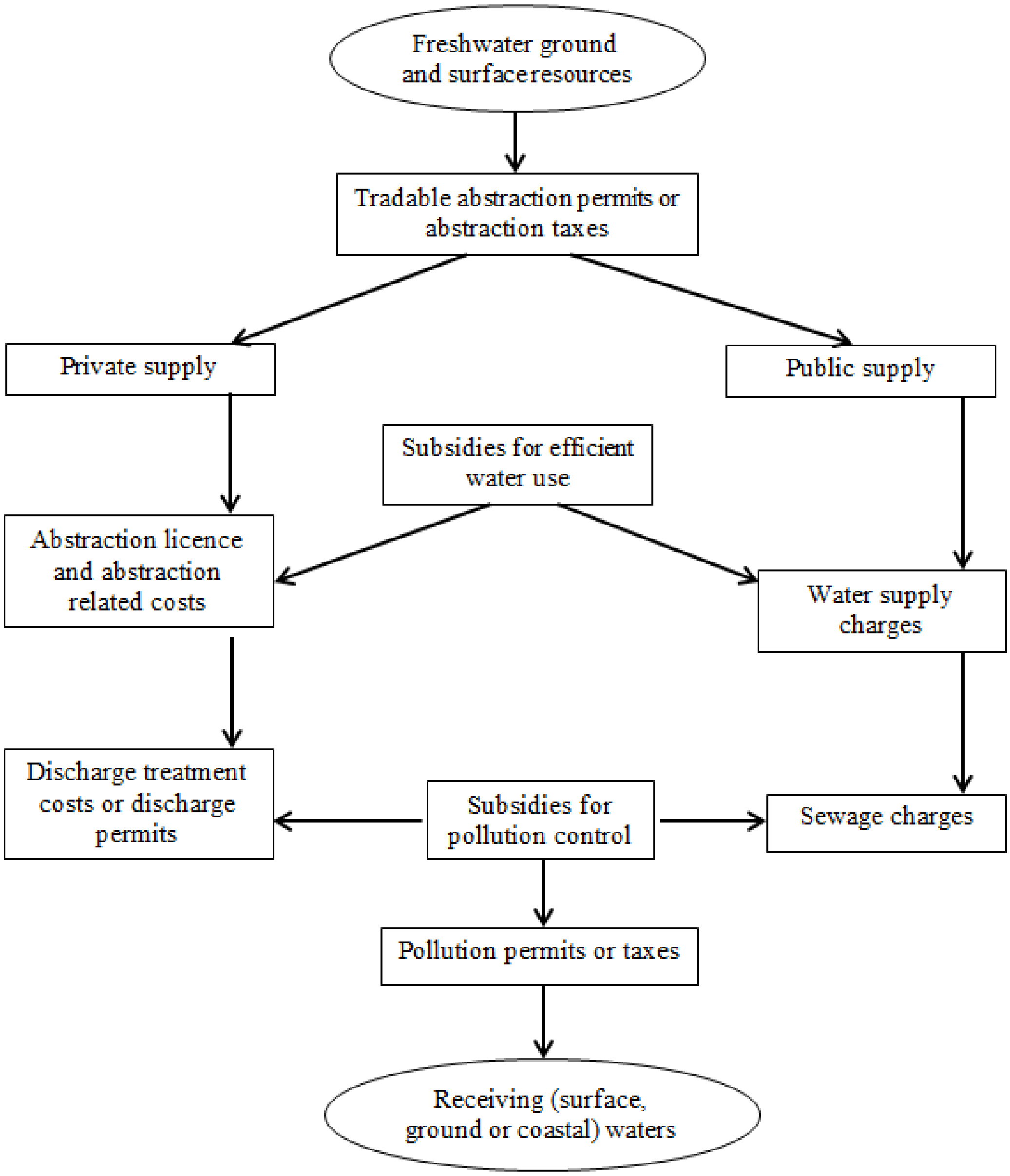
Figure 6.2. Priority catchments in Scotland for the first (2009 –2015) and second (2015–2021) RBMP cycles (DPMAG 2011).

The priority catchments are predominantly located in the East Coast and Borders areas where intensive agriculture takes place (Figure 6.2) and coincide with NVZs. All rural land users have the responsibility to adhere to the diffuse pollutions GBRs to reduce the deteriorating impacts on water quality of agricultural activities, such as storage and application of fertilisers, keeping of livestock, cultivation of land, discharge of surface water run-off, and application of pesticides (Scottish Parliament, 2008, 2011).

The diffuse pollution strategy in Scotland can be criticised for neglecting the synergies and feedback loops between environment, policy and economic activities that cause pollution at a catchment level. For instance, it currently lacks inclusion of economic instruments beyond economic support and incentives such as cross-compliance, funding from the Scottish Rural Development Programme (SRDP) and the restoration fund. However, paying farmers directly through these incentives to stop polluting might be unfair to farmers that are already not polluting as it means rewarding the polluters (Shortle, 2012). In addition, the cost of pollution control for the regulators would not be reduced as it would still be necessary to monitor whether the incentivised farms are no longer polluting and to identify the polluting farms. Inclusion of market based instruments (MBI) in the strategy could help achieve better results in pollution control and reduce monitoring costs while acting as an additional support for the farmers.

6.3 Role of market based instruments in diffuse pollution control

MBI are applied to the management of water (and other environmental resources) with the aim of complementing traditional policy options in accomplishing a certain policy goal, such as increasing environmental quality or promoting an efficient allocation of water among its users. There are various types of MBI applied to the management of surface and ground waters to introduce a higher economic efficiency (Figure 6.3).



*Charges are assumed to include relevant taxes.

Figure 6.3. Economic instruments used in water management (adapted from Kraemer et al., 2015).

Along with various air pollutants, including greenhouse gases, water quality trading (WQT) has been one of the areas in which environmental markets have been implemented. In pollution trading schemes, the regulator identifies the total pollution mass that can be disposed of at the source based on the highest concentration acceptable in the receiving water body. This cumulative load is distributed to polluters in the form of permits to pollute (Kraemer et al., 2015). The polluters are expected to trade their allocated permits to the point where the marginal value of each permit is equal and the permits are optimally distributed. The concept of WQT is applied not only to achieve economic efficiency and

environmental pollution control outcomes, but also to present an opportunity for buyers and sellers to reduce their pollution control cost via buying additional permits or to create extra profit through the sale of unused permits.

Over the last decade, a total of 26 WQT programmes have been established in the USA, the Netherlands, Australia, Canada, New Zealand, countries surrounding the Baltic Sea (Greenhalgh and Selman, 2012) and UK while others are under development. Although most WQT programmes address trading among point sources, some schemes are between point and agricultural non-point sources and between non-point sources.

Despite the advantage of delivering environmental improvements and economic efficiency of water-pollution control, WQT schemes have to be designed and implemented sensibly in order to account for the hydrological and geological conditions and economic and regulatory challenges at the local level. Several challenges are common to WQT schemes addressing diffuse pollution. Due to the transport coefficients and delay times of the pollution, the introduction of the loads and observation of pollution concentration is not easily and quickly linked to non-point sources (Shortle and Horan, 2008). Moreover, the burden of participation might be discouraging for small traders (Ribaudo et al., 2009). Therefore, depending on the feasibility, the regulating authority may take on additional administrative duties to reduce transaction costs for the individual traders. Another important issue is to establish trust and communication to increase participation by farmers in trading (Breetz et al., 2005). If farmers are already incentivised through payments for better land management practices, they should be convinced of higher benefits that they will get through participating actively in trading. When this is not achieved, WQT schemes tend to remain as “thin” markets where not many trades take place (Shortle and Horan, 2008).

6.3.1 Application of payments for ecosystem services to diffuse pollution

Payments for ecosystem services (PES) are economic arrangements used to reward the conservation of ecosystems through a series of payments to land or other natural resource managers to guarantee the provision of ecosystem services that originally come for free to those who benefit from the service. The novelty of PES compared to most MBIs arises from its focus on the ‘beneficiary pays principle’ (Smith et al., 2013) as opposed to the former ‘polluter pays principle’ that dominated pollution control policy.

The categorisation of PES schemes and environmental markets based on cap-and-trade are controversial in the academic literature (Pirard et al., 2010; Pirard and Lapeyre, 2014). Most scholars accept PES as an overarching term for a broad range of market-based conservation incentives based on Coasian discourse (Wunder, 2005; Engel et al., 2008; Gómez-Baggethun et al., 2010; Muradian et al., 2010). However, some argue that there is a fundamental distinction between PES schemes and environmental markets, as in markets actual property rights, e.g. fishing permits or water abstraction rights, have to be exchanged (Karsenty and Ezzine-de-blas, 2016). However, in market-based schemes for environmental services,such as clean water or carbon capture, that could be supported and improved by appropriate human activities, the buyers of the permits are paying to restrict sellers' full right to use their land rather than buying or leasing the property rights (Karsenty and Ezzine-de-blas, 2016).

Thus, water quality trading markets can be categorised as outcome-based PES schemes (Zabel and Roe, 2009; Defra, 2010b; Osbeck et al., 2013; Hejnowicz et al., 2014) where the prices are decided in reverse auctions and the bids are made online simultenously and competitively (Haruvy and Jap, 2008; Lundberg et al., 2016) before any contracts or transactions take place. Examples of PES in the form of water quality markets to concentrate on only non-point sources to reduce diffuse nitrate pollution are the Taupo Lake trading scheme in New Zealand and the Fowey River Improvement and Poole Harbour PES projects in southern England.

In the Lake Taupo trading programme in New Zealand, the Lake Taupo Protection Trust was assigned a NZ$ 81.5 million public fund in 2007 with the aim of achieving a 20% permanent reduction in nitrogen loads. In this scheme, polluters are allocated initial nitrogen load allowances based on their long term land use patterns and those who wish to discharge more than their allocation must acquire additional permits from others (Duhon et al., 2015).

The Fowey River Improvement Auction implemented in 2012 was the first example of a PES auction in the UK. Farmers in the Fowey River catchment bid to make land use changes that would improve the water quality, rather than trading pollution permits directly. The proposals that provided higher reduction potential than others were awarded funding by South West Water, the relevant water utility company, who would enjoy reduced treatment costs as a result of improved water quality (Day and Couldrick, 2013).

The Poole Harbour catchment scheme is based on a reverse auction between diffuse polluters (farmers) and point polluters (sewage treatment works, Wessex Water utility company and factories) with the aim of achieving 20 tonnes of nutrient, in terms nitrogen load, removal from diffuse sources, 80% of which is agricultural related. This target is divided into the trading units of 35 kg nitrogen load removal per hectare (ha) across a standard area of 25 ha. Once a farmer agrees to comply with the 35 kg per hectare reduction, they have to implement a set of measures that would result in this reduction on the land and provide evidence of implementation of required measures so that this allowance can be sold online to the highest bidder (RSPB, 2013) through an online platform.

PES incentives for sustainable land use have also been practised between Scottish Water and Scottish farmers (Scottish Water, 2013). However, no auction scheme has yet been applied to a Scottish catchment.

6.4 Methodology

In this analysis we adopt the principles of an environmental market based on two major features of water quality markets outlined earlier in the text.

First the buyers pay the sellers for the actual reduction of pollution they can evidence, not the good land management practices (not function-based payments but outcome-based payments) that are anticipated to improve water quality. The farmers have to prove that they have already achieved a certain reduction of nitrogen input per hectare of their land to create allowances (reverse-auction) before they can participate in the trade. Nitrogen use allowances for sale are calculated simply by the total difference between allowed nitrogen input per hectare and the residual nitrogen amount for the choice of agricultural activity in the current framework. Farmers who wish to sell these nitrogen allowances that remain from the initial amount they are allocated per hectare have to create them by switching from more polluting activities or crops to less polluting ones with smaller residual nitrogen figures.

Secondly, once the permits are created, rather than bilateral or multilateral negotiations and agreements, the unused allowances are put on the market without the identity of the sellers and buyers. An optimal market price for the permits is found as a product of availability of allowances and demand from the buyers by an online trading algorithm.

The rest of this section describes the formulation of a deterministic mathematical model for diffuse pollution control and welfare maximisation in the agricultural catchments. The model determines optimal nitrate loads for farms, enabling sellers and buyers to be defined by the difference between initial and the optimal distribution of N loads and the cost of abatement on each farm.

The model takes into consideration different profit scenarios on each farm based on various agricultural production options and their corresponding pollution outcomes at different groundwater sampling points. Index i ($i = 1,...,n$) is used to denote the farms as sources of pollution. Index j ($j = 1,...,m$) denotes the receptors, boreholes where the groundwater is sampled for water quality measurements. The portfolio of agricultural activities that take place in the catchment is denoted with a (a = 1,…..,A). Index t (t=1,….., T) denotes years. However, the current model is static and only considers a single year time frame, t=1 with single application of fertiliser and single round of auction.

The model considers the problem of duality from the perspective of a policy maker who wants to reduce the cost of pollution control while maximising the profit made at the catchment scale, rather than on an individual farm. The difficulty is achieving the ambient standards while maximising total profits for polluters (farms). The goal function (Equation 6.1) is formulated as a profit (welfare) maximisation function, linking nitrogen load with profits made from certain activities on each farm *i*.

$$max \sum_i \mathrm{f_i} = \sum_i^n f_i x_i \ ,\forall \mathrm{i} \tag{6.1}$$

Where variable x_i denotes the total amount of N load (kg) resulting from agricultural activity a on farm *i*. and $\forall i$ represents on each farm. The profit is linked with pollution output. A further elaboration of the profit function (Equation 6.2) is as follows:

$$max \sum_i \mathrm{f_i} = \sum x_{ia} \ \frac{p_a}{\gamma_a}, \forall \mathrm{i} \tag{6.2}$$

Where parameter $x_{i,a}$ characterises the resulting nitrate load from the production of option a on farm i, γ_a indicates the N load in the literature for the activity a and defined by the legislative kg/ha N application norms. Parameter p_a represents the profit made from each hectare allocated to option a on farm *i*. We assume that each farm i has A (a = 1,…..,A) options for agricultural activities, which yield marginal profits $f_{i,a}$ under total nitrogen load not exceeding legislative limits at catchment level.

We assume farms follow historical land use allocations and allocate the available agricultural land to each agricultural activity accordingly and in a way that is at least equal to the current allocation. This is to limit farms from shifting to only the most profitable activities and ending up in a monoculture that would be unrealistic in practice. These allocation ratios are replaced with crop rotation ratios when the model is upgraded to a dynamic setting that considers land use patterns over the years. Equation 6.3 is a provisional constraint.

$$\sum_{ia} \frac{L_{ia}}{L_i} \geq \ \alpha_{ia}, \forall \mathrm{i} \text{ and } \forall \mathrm{a} \tag{6.3}$$

Equations 6.4 and 6.5 depict two initial constraints, on land availability and nitrogen load, respectively. Where variable L_{ia} represents the land allocation to agricultural activity a on farm *i*, L_i represents the total amount of agricultural land on farm *i* and α_{ia} historical land use ratios for each activity a for each farm i. The first constraint is the land availability at farm level.

$$\sum_a L_{ia} \leq L_i \; ,\forall i \tag{6.4}$$

The total of the areas allocated to each agricultural activity on farm *i* cannot exceed the total available land for agricultural use on farm i. For simplification, it is assumed that the soil capability is the same throughout each farm and within the case study catchment.
The second constraint is imposed on the total N load at farm level.

$$\sum_i \bar{x}_i \; L_i \geqslant \sum_{ia} x_{ia} \; L_{ia} \; ,\forall i \tag{6.5}$$

Where $\bar{x}_i$ indicates the optimal pollution load on farm *i* and x_{ia} indicates the total actual pollution load created on farm *i*, both in kg. The total amount of N pollution created by the sum of agricultural activities can not exceed the farm pollution budget constructed by multiplying optimal pollution load and the total area of the farm *i* available for agriculture. The duality theory allows the price of environmental resource, here the pollution assimilation capacity of groundwater in terms of pollution permits, to be defined. Thus, Equation 6.5 can be further elaborated as below (Equation 6.6), assuming only a certain part of the pollution load on farm *i* reaches the receptor within one year:

$$\sum_i \; x_{ia} \; \mu_{ij} \leq \; \sum_i (\bar{x}_i \; + q_i), \; \forall i \tag{6.6}$$

Where q_i indicates the amount of additional N load allowances farm *i* has to buy or sell in order to maximise its profits and μ_{ij} represents the transfer rate between farm *i* and receptor j. For simplicity, here only one receptor is used and, to create the “thickest” market, the most downstream receptor in the catchment is chosen so that all the farms in catchment can be considered.

The change in the amount of pollution allowances on farm *i* after the trade might be negative or positive. While at farm level, the permits can change after trades, the number of allowances in kg bought and sold must be equal at the catchment level so that the catchment N load remains capped at catchment level. It is described as follows (Equation 6.7), where

pp_i represents the amount of pollution allowances (permits) bought and sold by farms across the catchment:

$$\sum pp_{pi} = 0 \text{ ,} \forall i \tag{6.7}$$

Appendix F summarises the description of parameters, scalars and variables calibrated for the model with their relevant units. The model is optimised using GAMS software in linear programming, first deterministically. The ultimate goal is to run the model stochastically, placing stochasticity in the load to concentration ratio resulting from groundwater transport and using several receptors along the catchment to highlight the link between where nitrogen pollution is loaded and its contribution to the nitrate concentration in the receiving water body. The GAMS code for the deterministic mathematical model is shown in Appendix G.

6.4.1 Case study and data used in model calibration

The Lunan case study catchment is located in the Angus region on the east coast of Scotland and has important natural assets such as the Lunan dunes and St Cyrus National Nature Reserve. The Lunan Water, the main water course in the catchment, drains an area of 134 km^2 from its source near the town of Forfar to the North Sea at Lunan Bay and is a lowland agricultural catchment with most of the area lying along a flat broad valley. Average annual rainfall for 2000–2009 was 890 mm and is quite uniformly distributed throughout the year. The groundwater in the catchment has special importance to the local community, meeting 50% of the water demand, representing a considerably higher dependence on groundwater than the average situation in Scotland. Porous bedrock underlies the catchment which makes groundwater bodies vulnerable to nitrate pollution (Dunn et al., 2014).

Nitrate pollution in the catchment has been and still is an important issue, especially in the groundwater which is affected by leaching from fertiliser use to enhance crop growth and the manure arising from livestock. Over the last decade, there has been a significant reduction in the nitrate concentration measured at the main surface water outlet before the sea at Kirkton Mill and the target concentrations for drinking water have been met (UKTAG, 2012). This indicates that the nitrate related environmental risk for the Lunan Bay estuary is also reduced. However, nitrate concentrations measured in groundwater are still slightly above the drinking water standard for nitrate-N (11.4 mg/l) both at Kirkton Mill and Balgavies measuring stations (Stutter et al., 2014), likely as a result of factors, such as lag times, mixing effects and biogeochemical processing in the soil (Dunn et al., 2014). Thus to address the current state of the nitrate diffuse pollution problem, the trading model here focuses on improving the groundwater quality rather than surface water quality.

The model was calibrated using the 2014-2015 Scottish agricultural census data for land use, land availability and for the location of the farms. Per hectare or per animal yields (in tonnes) and profits (in £) of the agricultural commodities are calculated using the statistics from the 2014 edition of the SRUC farm management handbook (SAC Consulting, 2016). Average ratios of N remaining in soil as a result of cultivation of a certain crop and annual manure output per different type of animals were compiled from the literature (Thompson et al., 2007b; Knight et al., 2008; Defra, 2010a, 2011). These are assumed to be the same throughout the catchment as the model produce results for the whole catchment, not for each sub-catchment.

The upper supply limit of each commodity (crops and livestock) is calculated based on the availability of the right type of land and per hectare, assuming a maximum output if all available land in the catchment is allocated to this single crop or livestock. The distances between farms and receptors are calculated in ARC GIS using postcodes from the Scottish Agricultural Survey June 2015. The pollution transport related estimations can further be improved by coupling the soil permeability map of the catchment (Lilly, 2017) in GRASS GIS and be further complemented with assumptions based on the SEPA groundwater report (Feuvre, 2010) and the findings of the NIRAMS model (Pohle and Gladwell, 2017). However, due to unavailability of NIRAMS test results at the time of modelling and later time constraint this could not be done in the scope of the current study. Such analysis has to be run separately for cultivated (arable) and semi-cultivated (grazing and rough grazing) areas (or the scenarios) of the farms separately and be included as a parameter for each farm in the model.

6.5 Results

The optimisation model aimed to achieve an annual 40% reduction in the agriculture related nitrogen pollution in the catchment. The initial share of agriculture in nitrogen budget is calculated using the agricultural sources figures in the SEPA/ADAS database for the Lunan catchment (Scottish Government, 2009b). A 40% annual reduction aim was selected because the prevailing scientific opinion suggests that a reduction rate below 30% might not be sufficient (Duhon *et al.*, 2015) to achieve significant reductions and the findings of the STREAM model recommend aiming for 10% below the EU legislative concentration in the Lunan catchment (Dunn et al., 2010). The aim is implemented by limiting per hectare load of nitrogen to 75kg.

Assuming that less than 60% of the N input reaches the measurement point at the end of the first year (Dunn et al. 2010), a rough replacement ratio of 0.5 between reduction at source and reduction at the measurement point within the same year is accepted. This is also consistent with the 2:1 ratio between diffuse and point pollution sources advised between diffuse and non-diffuse polluters in trading schemes (Shortle, 2013). Therefore, when calculating avoided treatment costs it is assumed that 2 kg of nitrogen has to be reduced at source in order to compensate for the 1 kg of nitrogen reaching water treatment facilities.

Findings of the deterministic optimisation model (under these assumptions) are listed in Table 6.2 with the figures from the diffuse pollution schemes in Poole Harbour and Lake Taupo for comparison.

Table 6.2. Comparison of figures from different diffuser pollution trading schemes.

	Location		
Gains from trading and type of scheme	**Poole Harbour, England**	**Taupo Lake, NZ**	**Lunan Catchment, Scotland**
Cost of nitrogen reduction (£/kg N per year)	£2.00	£1.70-£2.20	£1.40
Nitrogen savings (in % and in tonnes of N)	30%*, 47.5 t (Mann, 2016; Elliot, 2017)	20%, 183 t (Duhon et al., 2015)	40%, 102 t
Avoided fertiliser cost (£)**	10,160	46,482	25,908
Avoided treatment cost (M£)***	0.2-2	0.91-9.1	0.47-47.5
Type of trading scheme	Reverse auctions	Cap and trade with initial allocation and buy-outs via the Catchment Trust	Cap and trade with initial allocation and auctions
Targeted receiving water body	Rivers	Lake	Groundwater

*The annual reduction through trading is 47.5 t N in the catchment rivers. However, this does not account for the removal in the groundwater and other water resources. The targeted reduction for diffuse pollution by farmers is around 32% (500 t N) of the whole annual N budget in the catchment (Mann, 2016).**Cost of fertiliser is assumed to be 254 £/tonne for 20:10:10 NPK fertiliser composition aggregating the average 2017 market prices (AHDB, 2017). Avoided fertiliser costs are calculated based on this unit price and N savings. ***A lower and an upper boundary for treatment costs are estimated based on figures reported by Wessex Water for Poole Harbour Catchment (Elliott, 2017) and each kg of N savings.

The model forecasts that a 40% reduction can be achieved at £91,000 a year, at £1.40kg / N, which is reasonable but low compared to the other schemes. Based on the limited publicly available information, bids in Poole Harbour scheme are estimated around £2kg / N(Wessex Water, 2015; Mann, 2016; Skellett, 2016) and Taupo Catchment trust pays £227-170/kg for permanent removal of N load (Duhon et al., 2015), equal to £2.20-1.70/kg N in annual terms, assuming N stays in the soil for 50 to 100 years and fertilisers are used twice a year. The difference in figures between the locations might be because groundwater is not as widely abstracted for public supply as surface waters by the water industry, as little as 5% in Scotland (SEPA, 2011a), therefore reduction in its quality might not be realised explicitly as in surface waters.

The average cost of nitrogen removal from the receiving water resources ranges between £10 and 100/kg N depending on the nature of catchment management or conventional treatment measures implemented (Elliott, 2017), both of which are costly compared to £/kg market prices found in all trading schemes. Furthermore, treatment costs are not static. Reducing water quality and increasing demand is expected to require significant costs in capital investment and operating costs in the future. Approximately £316.5 million per annum was spent for reducing nitrate concentration in all mains water supply between 2005 and 2010 in UK (Defra, 2007). By 2027, the volume of groundwater use, and thus demand for its treatment, in the UK is forecast to double from its 2003 level (UKWIR, 2004).

While costs to the water industry are significant, the overall social cost of nitrogen pollution in the soil is higher than the actual water treatment costs and potentially increased water rates. In the EU nitrogen pollution from agriculture is estimated to cost up to £112 billion per year, with the cost of damage to human health and the environment estimated to be up to £24/kg (UKGW Forum, 2017). This indirect cost reflects the value of well-functioning ecosystem services.

Trading makes farmers realise the full cost of their nitrogen use and has proved to provide a more economical way to reduce pollution costs and the need for further remediation. European Commission research found that if farmers were to pay the full costs of nitrogen pollution resulting from their use of synthetic fertilisers, nitrogen pollution would be expected to fall by 30% (European Commission, 2013). This finding is also consistent with the reduction target set here.

Furthermore, with the uncertainty of Brexit and climate change, the income from trading pollution can provide additional support to farm businesses. The price of permits estimated at £1.40/kg adds up to an average of £3500 additional income per year for a farm with 100 ha land, assuming the trades takes place twice a year in January and June before the fertilisers are applied.

6.6 Conclusions

Throughout the EU, diffuse pollution from agriculture is one of the main reasons for the poor quality of water resources (Spiller et al., 2013). Much of the nitrogen pollution in rural catchments is associated with the use of fertilisers to achieve higher yields in crops and with production of livestock. However, meeting the rising food demand of an increasing world population still makes it necessary to use nitrogen fertiliser. Subsequently, there is now a great requirement to improve the output efficiency of nitrogen producing processes to avoid environmental damage and related costs. This chapter proposes how PES can help incentivise a behavioural change in farmers for more efficient use of fertilisers and better pollution management on land through the adoption of a water quality trading mechanism. The results here have two main implications.

Firstly, environmental markets as MBIs have potential to add to the current policy mix for pollution control and water management. The reduction in nitrogen load does not necessarily mean reducing agricultural profit as proved by the optimisation model using historical land use patterns. Using trading schemes is a powerful tool to maximise the profit at the catchment level from the capped pollution by shifting it from low value to high value users. Through trading, farmers realise the full cost of their production and are encouraged to change their ways of production to reduce the nitrogen input. The water industry is a clear beneficiary of water quality improvements and can share the costs of pollution control. Examples from the USA, Netherlands, Germany and France show that activities by water utility companies, with the primary aim of controlling pollution from agriculture at the source, provides a mean of avoiding water treatment costs and of contributing towards the implementation of environmental regulations such as the Water Framework Directive (Council of European Communities, 2000; National Research Council, 2000; Brouwer et al., 2003; Postel and Thompson, 2005; Heinz, 2008). Also, as water utility companies mostly deal with smaller parts of overall catchments, they have an advantage over centralised environmental agencies in that they are able to identify and address pollution closer to where it arises (Brouwer et al., 2003).

Secondly, market mechanism such as PES can complement direct payments to farmers in the form of subsidies. The cost of nitrogen pollution reduction through changes in land management, manure storage and fertiliser application is estimated to be up to 10 times cheaper than removing nitrate from polluted water (European Commission, 2002). These good practices can be incentivised and financed through trading which makes the price of ecosystem services dynamic. Many farm businesses in Scotland are uncompetitive and unprofitable in the absence of subsidies (Scottish Government, 2010). Brexit can present an opportunity to reform the current subsidy schemes and replace direct payments with conditional payments based on environmental outcomes so that with the same investment farmers would be financially supported and the environmental quality could also be improved.

The proposed scheme is additionally significant for merging the findings of science with the economic tools to improve our understanding of how the fate of pollutions in the soil can be considered in water pollution control. Although the model has provided preliminary results, it has limitations and additional research would be useful to improve it and other similar trading schemes that include non-point resources.

First of all, stochasticity has to be incorporated into pollution transport in the groundwater, agricultural commodity prices and yields. Different land management practices should be factored in as a parameter in the model. The effects of extreme weather events have to be accounted for, especially floods and their flushing effect on transport and concentration of the pollution in soil and sediment transport. Further modelling improvements include dynamic modelling of pollution load over the policy period of 50 or more years with bids, possibly twice a year in June and January, before the planting season. It is also necessary to consider crop rotations as a constraint and maximise farm profit over these periods rather than per year or harvest. The general trading framework could further be complemented by extensions such as coming up with an initial allocation of permits based on incentivising other outcomes, such as diversification or competitiveness on farms. Finding the trade-off between reducing the transaction cost for individual farmers to incentivise trading and the overall administrative cost of the trading scheme to the regulatory body or/and the beneficiaries would be a further possible improvement to the model.

Chapter 7 Conclusions

7.1 Contribution of the research

Water has an economic value in all its competing uses and thus should be recognised as an economic good (ICWE, 1992). While many countries recognise the vital nature of water resources, few, if any, pursue a rigorous analysis to reveal the explicit value of water as a basis for determining whether water is actually being allocated to sectors in order to maximise its overall benefit to society.

Water in Scotland, like elsewhere, is allocated for a range of reasons that have little or nothing to do with optimising resource value and much to do with historical riparian rights attached to land rights. How should any country rationally consider the allocation of water resources when there is competition? This is an important question for a country that aspires to be the first Hydro Nation and water economy in the world (Scottish Government, 2016a).

This thesis analysed the status of water use and valuation, focusing on key Scottish industries, and investigated what can be done at catchment level to improve water management and overall economic benefit in Scotland. Efficient and equitable ways of re-allocating the rights to use and pollute water in the form of trading is explored to create the water economy of the future under changing demand dynamics induced by climate change. It addressed general gaps in the literature and contributed to the strategic aim of maximising the economic value of water in the Hydro Nation Agenda (Scottish Government, 2016a).

The conclusions of the thesis can be separated into two main categories: academic contribution and policy recommendations.

7.1.1 Academic contribution

This thesis explored the theory of economic efficiency of water use and how this theory and the social value from water can be maximised. There are two components of economic efficiency when it comes to water use. The first is the valuation of water in its competing uses. This is important to reveal the opportunity cost of its allocation to a certain use over another. The second component is the tradability of water rights so that a more socially optimal allocation can be proposed once the opportunity cost of current allocations is revealed. The overall academic contribution of this thesis is to provide an example of how the theoretical principle of economic efficiency of water use mandated by the WFD could be applied to water pricing in Scotland.

To this end, the thesis sets out to challenge the current allocation of water use rights and including the use implied by permissible water pollution, which are not necessarily distributed in line with economic efficiency. The analysis reveals the extent to which an economic efficiency principle applies to Scotland. Chapter 2 provides an overview of different water demands and their values in Scotland. The analysis uses statistical methods and makes assumptions based on for each water use to consider the type (at random or not at random) of missingness to patch data gaps in abstraction returns. It uses the resulting figures to account for current state of volumetric water allocations appropriately. Paired with the valuation estimates for each use, the analysis provides a framework of how a country or river basin scale portfolio of water uses can be constructed.

Following the analysis of the status quo of water allocation in Scotland, the thesis considers the under-reported topic of industrial water use. Most developed countries allocate a major percentage of their water resources to manufacturing (Jia et al., 2006; Scheele and Malz, 2007),but research on the valuation of this use is limited. Due to time limitations and unavailability of a single representative study, a meta-analysis was conducted to derive a transfer value from the available literature. The meta-analysis has two important academic contributions.

First, the use of meta-analysis is not widely employed economics literature, compared to medical and biological sciences (Ioannidis and Roberts, 2018). However, there is a need to provide more statistically robust estimates to be used in value transfers, to evidence hypotheses and to systematically synthesise literature in social sciences (Barnett-Page and Thomas, 2009; Brander et al., 2012; Davis et al., 2014). This need has recently increased the popularity and the application of meta-analysis to topics in resource management and economics with more than 35 peer reviewed studies published in the last 20 years.

Specific initiatives like The Berkeley Initiative for Transparency in Social Science (University of Berkley, 2016) aims to provide relevant training to researchers to popularise the adoption of the technique. This thesis provides the first meta-analysis that attempts to understand the valuation and price responsiveness of water uses in manufacturing (and extractive) industries. It contributes to the on-going development of meta-analysis in economics literature.

Secondly, apart from confirming the initial observation that there has been limited research on the the topic so far and a lack of representative studies to transfer values for Scotland, the meta-analysis provides an insight into the dynamics that influence industrial water use' globally. The findings inform pricing decisions relevant to commercial water use where the nature of water as an economic good is at its highest, both as a direct input and an intermediate factor of production. There is a risk of pricing commercial water use disproportionately if the factors influencing responsiveness of user demand are not considered. Such prices can induce undesired adverse effects on regional economies (Whittington, 1992) while trying to achieve economic efficiency in water use across uses.

The analysis is also important in linking the portfolio of water uses in Chapter 2 with the analysis of water use in the production of Scotch whisky, one of the highest-value creating and most water intensive manufacturing industries in Scotland, covered in Chapter 3.

A wide range of methods have been developed over time for the valuation of water use. However, we wanted to examine valuation in relation to pricing which is central to incentivise efficiency, sustainability and accountability in water use. Therefore, two different methodologies for two different sectors (manufacturing industries and livestock) are employed due to the nature of demand. While for the whisky industry we adopted the marginal productivity approach in Chapter 4, for the livestock sector we employed net back analysis in Chapter 5.

Value of water use in manufacturing and extractive industries are analysed using marginal productivity technique. The particular method is chosen to assess how dependant the production in whisky industry on water supply (output elasticity) is. This information is further combined with how much scope there is to improve water use productivity through demand-responsive pricing (price elasticity). The figures estimated for price and output elasticity, -0.8 for the overall food and beverage industry and 0.56 for Sthe cotch whisky industry, respectively, in Chapters 3 and 4. Both figures are comparatively high, making whisky dependant on water for the final output and comparatively inelastic to changes in this pricing.

A price elasticity analysis specifically targeting the Scotch whisky industry is expected to be even higher due to the specific position of whisky as a premium product that is protected with geographical indication (citation) within the beverage industry. The elasticities indicates that pricing at the margin rather than by access to water through abstraction licences is a possibility for industries that produce high-value added outputs. Such implementation will encourage sector wide technological improvement in terms of water efficiency (Garcia and Reynaud, 2004). To our knowledge, this type of diagnosis based on analysis and synthesis of overall and sub-sectorial industrial water demand dynamics had not been reported in the literature at the time of thesis preparation.

To analyse water use in the livestock industry, whilst the marginal productivity technique is also applicable here, we used netback analysis. The choice is based on our interest in measuring livestock farmers' ability to pay (ATP). When analysing agricultural water demands the willingness to pay figures might not always be reliable in determining what farmers are able to pay in reality. This is mostly because there are variations in characteristics that determine the individual farmer's willingness (Ndetewio et al., 2013). Even in a developed country setting, as in the case of Scotland, farmers must address a complex set of decisions when selecting optimal farm management strategies and might not be fully aware of their water productivity (Wichelns, 2014). Also smallholders of most farm types, especially those in upland and hill farming require subsidies to remain in business and face challenges of profitability. Therefore, the question of their ATP is a more relevant measure for the economically efficient yet affordable prices that the majority of livestock farmers can actually pay (Njoko and Mudhara, 2017).

While the results here are open to improvement,with this analysis, we provide the only available water use valuation estimate in livestock industry where the amount and consequence of water use has caused a lot of global debate. Currently up to 30% of the total water footprint of agriculture originates from the production of animal products. This figure is expected to increase with rising global meat consumption and the intensification of animal production systems (Mekonnen and Hoekstra, 2012).

The water abstraction volumes related to livestock or overall agriculture may not yet be a major issue in regions where the agriculture is rain fed. However the uncontrolled diffuse pollution as a result of agricultural production is.The main problem is that due to pollution originating from agriculture, the cost of water use due to treatment costs are increased for all users as a negative externality. Therefore, in the tradability section we looked into the contemporary topic of payment for ecosystem services and how to design a scheme that implements the principles of economic efficiency to address the widespread issue of diffuse pollution control.

The aim is to propose a trading mechanism to transfer water use from the ow value users to high value users in order to increase the overall return from water pollution capped at a catchment scale. To implement such schemes it is necessary to incorporate the natural processes that deal with the fate of pollution in the soil into the economics of trading. Set in the context of Scotland, the model here can be calibrated according to any other country using relevant legislative pollution limits, land use, profit and soil type information.

This thesis makes a contribution by incorporating the findings of the previous models and literature and by translating pollutant loads in farm into concentration at groundwater boreholes where measurements are made. The general principles of ecological economics (Daly, 1991; Cavalcanti, 2010), and theoretical literature on water quality trading specifically (Ermoliev et al., 1996, 2000; Morgan et al., 2000; Horan et al., 2002; Kerr and Lock, 2008; Nguyen et al., 2013) dictate consideration of physical components in water quality trading as human-environment system as a whole. However, at the time of this thesis's preparation, as far as we know, there were no studies that had done so using real world data. In actual practice the implementation of this principle does not go beyond nutrient budgets. We used a real catchment as a case study and data from this catchment to calibrate the model. This applied aspect was also a novelty made in the scope of this thesis.

7.1.2 Policy recommendations

Positioned in the domain of applied (environmental) economics, the thesis looks to accomplish a more practical contribution rather than a theoretical one by applying existing methodologies to new areas. We focus on how the interdisciplinary research carried out here can provide implications relevant to both Scottish and wider European policy goals in water management. The policy-relevant conclusions of thesis are summarised below by chapter.

Chapter 2 examined the current allocation of water among its users and their valuation of water use by identifying the users, the volume of water they use and at what value in order to present an overview and identify opportunities for improvement in the social return. The analysis has three important implications.

The first is that more integrated accounting of water use is needed. The way water uses are classified differ between the two main bodies that regulate water use in Scotland. While

Scottish Water (SW), who provides mains water supply, uses SIC codes to categorise the non-household customers, SEPA, who issues the abstraction licences, categorises the licences based on the purpose for water abstraction such as irrigation or evaporative cooling. Since 2013, SEPA has adopted a practice for collecting voluntary data returns from abstraction licence holders; however the average percentage of returns in general has been low, around 20% for all uses and 0% for some sectors, such as hydropower. Gradually making this voluntary scheme obligatory for the registration of new licences and the extension of existing ones and using a joint system between SEPA and SW would greatly improve the ability to account for water allocation in Scotland at both catchment and national levels.

The second is that the trade-off between hydropower and environmental flow requirement requires more attention. Currently hydropower has the highest allocation among all uses in Scotland. This is expected as Scotland currently hosts 145 hydroelectric schemes producing approximately 12% of its current electricity (IH Energy, 2016). Despite its non-consumptive nature, hydropower schemes temporarily divert water from its natural course, depriving other in-stream uses of their share during the storage period unless it is a run-of-the-river plant whereby little or no water storage is required. In Scotland, hydroelectricity capacity is shared between conventional (68%) and pumped-up storage (32%) hydro schemes (Sample et al., 2015). Both plant types dam the water to produce electricity. If the environmental flow requirement is not considered in the planning process, certain elements of the ecosystem, such as fish and other aquatic life forms, which are sensitive to reductions in water flow and level in-stream can be permanently damaged. With its considerable potential for additional hydro schemes, Scotland is expected to increase its hydroelectricity capacity as part of the means to achieve its 100% renewable electricity target by 2020 (Scottish Government, 2017a). While further development of hydropower should be supported, the trade-off between hydro schemes and the water environment must always be considered in deciding location, capacity and type of hydropower scheme to be commissioned.

The third finding is that, with the exception of households, all consumptive users apparently pay much less than estimated here to be the value of their water use. Unsurprisingly, the most value from water use is created in service and manufacturing industries in Scotland. The gap between cost and value of water to the manufacturing industries reveals a foregone opportunity cost to the public. Though obvious, this information is important to consider when making allocations under competition at catchment level.

Considering the volumes consumed and values created in industries (other than service industries), the research suggests that there is a lack of appropriate pricing in industries. Chapter 3 estimated the value of water to its users in different industrial sectors and how elastic their water demands are in response to changes in price, as well as which factors tend to affect both estimates. Synthesising the available empirical literature on industrial water demand, the analysis revealed up-to-date figures for price elasticity and valuation of industrial water demand modified for the UK, as well as a meta-analysis of factors that are assumed to cause variation between different estimates in both. The analysis has two main implications.

Firstly, industries that use water only in production processes and cooling have relatively elastic demands in response to increasing prices. However, demand from industries that also use water as a direct input to production is less affected by changes in pricing. For the first group, there is a higher possibility to reduce water use due to a substitutability between the choice of technology or other production inputs without changing production levels. Thus, the effect of pricing on water conservation behaviour of such industries is expected to be greater.

The second finding is that industries have an average economic value of 3.6 £/m^3 for their water use, with each sub-type of industry having a different shadow price of its water use linked with the value created from this use. This should be further considered in allocation decisions under competition and when estimating the willingness-to-pay of industrial users for their water supply.

The following two chapters considered two case studies focusing on important sectors in Scotland where research is very limited. The studies highlight the potential to support and develop opportunities for industries for which water is a critical resource.

Being one of the most important manufacturing industries, whisky is Scotland's second largest export, after oil and gas (SWA, 2011). Increasing global demand for whisky (SWA and 4-Consulting, 2015) and projections for reduced seasonal water availability in several regions of Scotland (Brown et al., 2012) suggest water related pressures and risks to the industry. Chapter 4 contributed to the current knowledge on water use and its valuation by the Scotch whisky industry. A water footprinting (WF) analysis of the production chain and marginal productivity of water use in distilleries and other parts of the chain was conducted. Both analyses have important implications.

The WF analysis found that water is mainly embedded in the supply chain and the total WF of whisky comes from green water used in barley production. The WF_{green} can only be used by crops. Thus, its only opportunity cost is choosing to grow barley among available crop options. This makes the WF_{green} WF with the lowest opportunity cost. Although only 2% of the water use in terms of WF takes place in the distilleries, the distilleries rely on the quality and quantity of local water resources to maintain their operations. While Scotch Whisky Regulations allow for the import of spring barley, the manufacturing processes have to take place in Scotland for any whisky to be classified as Scotch (UK Parliament, 2009). The high value of water use in distilleries (£5.60/m^3) indicates that distilleries will be willing to pay premium prices to safeguard their water supply when future competition for water makes water markets at catchment level a feasible option to reallocate available water.

Chapter 5 explored the value of water use in Scottish beef and dairy production from the ability to pay perspective. The current price scenarios estimated for water supply highlight the gap between the price and value of water use on livestock farms assessed with netback analysis. Several important conclusions can be drawn from this analysis.

Dependency of the livestock industry on water supply is apparent. The results of the netback analysis show that the economic value of water use in livestock is as high as domestic and some industrial water uses, and much higher than the value transferred for agricultural irrigation in the Scotland-wide water valuation in Chapter 2 on the current allocation and valuation of water use in Scotland. The charges associated with the provision of mains water and licensing of private supplies partly reveal the cost of managing water from the supply side in compliance with EU WFD (European Commission, 2000). However, they do not reflect the full value of volumetric water use in line with economic principles. Being the first study to produce a £ per m^3 value for water use on livestock farms, the results can also be transferred to other parts of the world with similar climatic conditions and farming practices.

The apparent disparity between the cost and price of water in the livestock industry in Scotland indicates that reforming distribution of abstraction licences in a way to match the value would incentivise more sustainable water use options, such as rain harvesting and upgrading to recycling and reuse technologies on farms and other facilities.

In Chapters 2 to 5, the volumetric allocation of water among some users has been investigated, but water use is not always volumetric and pollution consumes water in the receiving water bodies by requiring volumes to dilute the concentration to an acceptable level (Hoekstra et al., 2011). In consultation with SEPA, the most policy relevant water issue in Scotland was identified, and how it could be improved by application of trading. Although higher competition is expected as an effect of climate change in the near future, at present rural diffuse water pollution caused by fertiliser use and livestock production is the greatest impediment to Scotland achieving full compliance with EU WFD standards (Scottish Government, 2015c). Pollution degrades the environmental quality of water bodies, increasing the treatment cost of water for all its users. Chapter 6 shows the design of a water quality trading scheme to control nitrate pollution in the groundwater. It combines basic soil dynamics with trading, which is novel compared to the few diffuse pollution trading schemes in practice. A trading model was developed, calibrated and tested for the Lunan case study catchment. The most significant findings from the modelling study are summarised below.

Firstly, as the result of trading the model forecast that overall income created at catchment level increases. This scheme can be extended to a variety of persistent and nutrient pollutants and water bodies and can provide farmers with additional income, partly replacing subsidies. This indicates the potential of market-based instruments, such as WQT trading schemes used in Chapter 6, to complement the current policy mix for water management by rewarding the conservation of ecosystem services and providing suggestions for reforming subsidies that are currently mostly in the form of direct payments to farms in Scotland (Kenyon, 2017).

Secondly, to implement such schemes it is necessary to incorporate the natural processes that deal with the fate of pollution in the soil into economic principles of trading. Otherwise, it might be problematic to define and regulate property rights in terms of allowances to pollute and to incentivise sufficient participation in the scheme so that the cost of pollution control will be efficiently reduced. Here, this thesis makes a contribution by incorporating the findings of the previous models and literature and a coefficient that translates load in each farm into concentration at boreholes where measurements are made in the trading.

7.2 Limitations of the research and future work

A dissertation is limited by the timescale. Plus external factors that can handicap application of chosen methodologies, including the availability and quality of data sets. The main constraint with this thesis has been the timely access to required data sets and the challenges of working across several disciplines, all of which slowed the analysis. Despite the detailed contribution to the literature and the contemporary environmental policy goals in Scotland, it is important to acknowledge shortcomings of the work.

In Chapter 2, there might be over and underestimation of the volumes allocated to different water uses due to unavailability of data returns on actual water abstractions. As already mentioned in Chapter 2, this situation can be improved with more systematic accounting of water use and regulations that would oblige the users to cooperate with the environmental regulator. The broad brush approach adopted in the valuation of water uses aims to inform a country-level situation in water allocation and pricing policy. Thus, the estimates might not be valid for each catchment. In turn, evaluating the value of allocation choices, such as leaving more water in-stream, would facilitate a more thorough and representative analysis at catchment level. Initially a catchment scale case study in collaboration with the Spey Catchment Initiative was considered to explore the value of environmental flow requirement in the catchment where environmental flow competes with two major value creating Scottish industries of hydropower and whisky. However, this could not be pursued due to time constraints. It could instead be taken up as a further research project in the follow-up of the PhD.

In Chapter 3, there is uncertainty surrounding the data collection of the primary studies, which provide the source data for the meta-analysis. An attempt was made to test the effect of this inherent problem of the methodology in the meta-regression which yielded insignificant results. It could be improved by contacting study authors directly. However, given the publication date of some of the primary studies, establishing this contact might not even be possible or worthwhile.

Chapter 4 on the Scotch whisky industry has limitations in its estimations, as assumptions are based on the limited publicly available data. Future research based on collaboration with the whisky and other industries could improve access to industrial data and facilitate expert consultation from the industry for more robust estimates.

The volatility of agricultural markets has made the valuation estimates for water use in the livestock industry in Chapter 5 not fully applicable to outlying years where agricultural prices were historically low, such as 2014 when UK milk prices dropped by almost 20% (Matthews, 2015). Future research could look into modelling stochastic commodity prices to replace the historical statistics used here, so as to improve the results with uncertainties related with farm profitability.

The diffuse pollution control model in Chapter 6 does not have a multi-year setting, which would enable the consideration of residual nitrogen in the soil from previous applications and the crop rotation limitations in profit maximisation. In addition, pollution related information is borrowed from the SEPA/ADAS database and partly from NIRAMS model which rather than being a groundwater specific tool, measures the combined effectiveness of diffuse mitigation measures and assumes uniform pollution at different locations and depths of the aquifer (Sample and Dunn, 2014). Leaching of nitrate in the soil could be more precisely modelled with MT3D, a modular three-dimensional multispecies transport model for simulation of advection, dispersion and chemical reactions of contaminants in groundwater systems, (Bedekar et al., 2016) and MODFLOW (Harbaugh, 2005) groundwater modelling software, a computer code that solves the groundwater flow equation to simulate the flow of groundwater through aquifers,

Future research could investigate how the precision of pollution trading schemes can be improved and how property rights related to pollution can be more clearly defined. In its current state, the model can be considered as a work in progress and the time constraints did not allow for the incorporation of the stochasticity, location of the load and other planned extensions in the conceptual model. With more time, the model could also be extended to explore further questions, such as the optimal levels of subsidies and diversification on farms to increase catchment level profitability and to identify how environmental policy goals and profitability of farms can be merged.

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Appendices

Appendix A: The justification of valuation methodologies chosen and their assumptions and shortcomings

Water use	Valuation method used	Justifications for the choice of method and, if relevant, of primary study	Key assumptions made and possible shortcomings
Domes-tic water use	Scottish Water charges for metered households	Water charges are assumed to reflect consumers' valuation (Young, 2005; Eftec and Ofwat, 2011) in the absence of other valuation figures.	It is assumed that the SW water charges are based on cost recovery principle of WFD and SW conducted a market-search before constructing them. Water charges are expected to be below the consumers' willingness and ability to pay (Tabieh et al., 2015) and SW average household charge is among the lowest in Great Britain (SW, 2016). For these reason, this figure can be interpreted as a lower boundary for metered households' valuation of their water use.
Manu-facturing industry	Meta-analysis	The unavailability of secondary data and time required to collect primary data made value transfer necessary here as in many other uses. There was no single primary study applicable enough to Scotland to transfer results. Therefore, a statistical analysis that synthesised all available studies was considered to be a more robust option (Bal and Nijkamp, 2001; Ioannidis and Roberts, 2018) for value transfer.	The primary studies pooled for price elasticity and monetary valuation sample indicate that this area of research has not fully matured enough to provide the type and number of studies required to conduct a roboust meta-analysis,
Service industry	Value transfer	The primary study chosen is a recent study that used a big sample of primary data collected from another developed and EU member country that implements WFD These features made it the most representative among the few available	Zaragoza implemented measures such as revising water tariffs to provide disincentives and incentives for water use efficiency and to ensure a full cost recovery through the Zaragoza Water Saving City programme starting from 1996 as a

		studies reporting a single value for the service industry. The significance of water to hospitality (Gössling et al., 2012; Gössling, 2015) and the significant share of the hospitality industry in the Scottish economy (Scottish Government, 2017d) also supports the choice of transferring the results of shadow pricing analysis by Angulo et al. (2014).	response to water scarcity (Smits et al., 2010). Similarly, Scotland introduced the first retail competition for non-household customers in UK in 2008 and established a competitive allocation of water in non-household water supply (Deloitte, 2017). Therefore, despite the certain climatic and water availability differences between Spain and Scotland, due to similarities in relevant policies and their implementation, the primary study is assumed to be relevant
Irrigation	Value transfer	The availability, recent date and Scottish scope of the primary study justifies this choice of value transfer (SNIFFER, 2005).	The value of irrigation can change with the efficiency of irrigation technology and choice of crops (Marchant et al., 2018). Irrigation , technology used is similar across the country (Knox et al., 2007) and potatoes cover the largest irrigated area in Scotland (Scottish Government, 2012). Therefore, no major shortcomings are identified.
Live-stock	Netback analysis	The water use on dairy and beef farms was estimated using netback analysis and statistics on commodity markets (SAC Consulting, 2016) as there were no available studies from which to transfer a value.	Only the Scottish Suckler Beef Scheme is assumed as subsidy as it applies across all regions. Also the commodity market prices are assumed to be stable although they are not in reality. This might make the findings less representative for years that have outlying market prices.
Hydro-power	Value transfer	The availability, recent date and Scottish scope of the primary study justifies this choice of value transfer. In the primary study the nature of the full economic costs of use is considered and a monetary value is constructed accordingly (MacLeod et al., 2006). The highest cost of water use stated is assumed to be the marginal value of	The value created at a hydropower facility is heavily dependent on the size and type of the facility and depends on which technology it is compared with (MacLeod et al., 2006). Thus there might be differences between hydropower schemes across Scotland.

		hydropower as it is still profitable.	
Naviga-tion	Value of boating licences per m^3 of water abstracted for this use	Navigation in inland water ways is mainly categorised into main two groups: commercial (passenger or freight transport)and recreational navi gation (boating, sailing etc.) (United Nations, 2004). The most common recreational navigation activity in Scottish canals is boating (Scottish Executive, 2011). Accounting for each recreational activity separately might end up in double counting. Thus, boating is assumed to be the most representative navigational activity in inland waters in Scotland.	The water abstracted for navigation via relevant abstraction licences might have additional in-stream benefits to maintaining navigational water levels.These positive externalities are not valued and discounted. On the other hand, boating is supposed to have some benefit to local tourism revenues which is also not accounted for due to the unavailability of relevant data.
Environ-mental flow	Value of fishing related revenues per m^3 of water abstracted for this use	The lack of representative primary studies for value transfer and data and time intensity of carrying out a complex primary analysis have led to adoption of the proposed methodology. We made three assumptions linking fish stocks with water levels: (1) healthy fish stocks are assumed to be the indicator of sufficient water availability averaged over the year and quality; (2) the availability of water is the primary requirement for the survival of fish stocks; (3) the River Speyis assumed to be representative of all the (salmon) rivers in Scotland.	The assumptions made might be problematic in several ways: Other impinging factors such as chemical composition of water, rainfall patterns, water temperature, water velocity that have impacts on critical fish habitats are not considered. The analysis is purely based on hydrological water availability and does not consider geomorphological and ecological components related to environmental flows (Smakhtin et al., 2004). Finally, the assumption is not dynamic in a way that considers seasonal availability of water and abstractions for competing uses, which requires location specific considerations.
Aqua-culture	Value transfer	The availability, relatively recent date and Scottish scope of a primary study has justify this choice of value transfer (SNIFFER, 2005).	No major shortcomings are identified.

Appendix B: List of primary studies

Price elasticity data set reference	Number of observations	Study code	Case study country
Babin et al., 1982	5	11a	USA
DeRooy, 1974	2	11b	USA
Dupont and Renzetti, 2001	3	11c	Canada
Ferres and Reynaud, 2003	5	11d	France
Frederick et al., 1979	1	11e	USA
Grebenstein and Field, 1979	2	11f	USA
Hussain et al., 2002	1	11g	Sri Lanka
Kumar, 2012	10	11h	India
Malla and Gopalakrishnan, 1999	4	11i	USA
Nahman and deLange, 2012	12	11j	South Africa
Onjala, 2002	8	11k	Kenya
Rees, 1969	5	11l	UK
Renzetti, 1988	2	11m	Canada
Renzetti, 1992	4	11n	Canada
Renzetti and Dupont, 2003	1	11o	Canada
Reynaud, 2002	15	11p	France
Rojas, 2005	7	11q	Mexico
Schneider and Whittlach, 1991	1	11r	USA
Tobarra-Gonzalez, 2015	6	11s	Chile
Turnovsky, 1969	2	11t	USA
Wang and Lall, 2002	15	11u	China
Ziegler and Bell, 1984	1	11v	China

Monetary valuation data set	Number of observations	Study code	Case study country

US Environmental Protection Agency, 2013	1	12a	USA
Fuji et al., 2012	5	12b	China
Ku and Yoo et al, 2012	22	12c	Korea
Kumar, 2012	10	12d	India
Nahman and de Lange, 2002	13	12e	South Africa
Rojas, 2005	5	12f	Mexico
Tobarra-Gonzalez, 2015	6	12g	Chile
Wang and Lall, 2002	16	12h	China

Appendix C: R code for meta-regression

The code can be found in the CD attached to the back cover of the thesis.

Appendix D: Classification principles behind industrial sub-categories

1. “All manufacturing industries” include observations made across the entire samples of primary studies.

2.“Chemical and allied industries” includes observations classified as “Chemical”, “Chemistry”, “Rubber and plastic”, “Petrochemicals”, “Drug and pharmaceuticals” and “Fertilisers” in the primary studies.

3.“Electrical and mechanical industries” includes observations classified as “Electric, electric, electronic, communication” “Transport equipment”, “Industrial equipment and machinery”, “Electrical apparatus”, “Precision equipment“, “General machinery”, “Electrical apparatus”, “Transport equipment”, “Electronic equipment, “Medical equipment” and “Automobiles and parts”. In some studies what has been classified here as “Electrical and mechanical” industry has been represented in more than one observation under classifications such as “Precision equipment” and “General machinery”. As these observations represented different data sets that are both relevant to this sub-category, they were kept in the sample and assumed as separate observations of “Electrical and mechanical industries” originating from the same primary study.

4. “Food and beverage industry” includes observations classified as “Food, beverage, tobacco”, “Food and beverage”, “Food”, “Food production”, “Sugar”, “Beverages”, “Alcohol” in the primary studies. Food and beverage might be categorised as one common industry or into two separate or even more specialised industries like “Alcohol” (Renaud, 2002), “Distilleries” (Kumar, 2012) or “Sugar” (Onjala, 2012). As these observations represented different data sets yet are all relevant to this sub-category, they were kept in the sample and assumed as separate observations of “Food and beverage industry” originating from the same primary study.

5. “Mining and allied industries” includes observations classified as “Energy resource extraction”, “Metal (metallurgy and steel)”, “Non-metallic mineral, primary industry”, “Steel and other metals”, “Non-metallic”, “Metallic”, “Coalmining”, “Smelting” “Metal fabrication”.

6. “Paper and allied industries” includes observations classified as “Wood, paper, publishing”, “Paper and paper products”, “Paper”, “Forestry and paper”, “Paper and allied industries”, “Paper and allied industries”, “Paper and wood”.

7. “Petroleum related industries” category includes observations from “Petroleum refining”, “Oil and gas”, “Oil” and “Petrochemicals”. It is also an option to further group “Petroleum related industries” and “Mining and associated industries” together to obtain a separate “Extractive industries” data sample.

8. “Textile and allied industries” include observations listed as “Textile”, “Textiles, apparel, leather”, “Textiles, shoes, leather”, “Leather” and “Spinning” in the primary studies.

9. “Unclassified manufacturing industries” is a miscellaneous group that includes “Others”, “Other industries” categories from primary studies. Moreover, broad observations classified as “Wood, chemicals and other products” that might be in more than one group but due to restrictions cannot be added to one are included here. “Household goods and textiles”, “Chemicals, petrochemicals, pharmaceuticals and cosmetics”, “Pharmaceuticals and biotechnology”, “Chemical, wood and other products”, “Chemical, petrochemicals, oil and gas, pharmaceuticals and cosmetics” do not fit in any single category described here. Thus, such observations were included in the “Unclassified manufacturing industries” and are regressed together with observations categorised under “Furniture and other manufacturing industries” in the primary studies. Two observations of “Construction” and “Construction and building materials“ were also included to this group as they were too few to constitute a separate group for meta-regression. We assumed these observations are for manufacturing of construction materials.

10. A few observations categorised as “Service Industries” and “Power” or “Energy Production” in the primary studies were removed as the analysis here targets manufacturing (and extractive) industries.

Appendix E: CD Pearson coefficients and collinearity statistics

The Pearson correlation coefficients show that water, energy and materials resulted in a correlation greater than 0.99 (Table 10). This indicates multi-collinearity issues in the model. Thus, it is not possible to use a Translog specification because applying a Translog production function would exacerbate collinearity already present in the data set (Pavelescu, 2011) and makes it further complicated to isolate the effect of each input variable on the final valuation and increases the standard errors. This issue arises because the data for energy and materials is directly derived from the water abstraction data.

Variance Inflation Factor (VIF) statistics for water, energy and materials are also critical (Table 9) as a result of how data sets on energy and materials are constructed nested on water use data. Thus, here CD functional form might also not be the most appropriate to represent the technology, energy and materials used in malt whisky distilleries with the available data set. The CD production function implicitly assumes that the Allen elasticity of substitution between all pairs of inputs has a (negative) unit value (Fuss and McFadden, 1978). Although water, energy and materials are theoretically perfect complements, in practice perfect complementarity of inputs is rather unlikely. For this reason, another production function with CD-Leontief form is applied to the data. The results for Pearson correlation and VIF collinearity tests for both production functions are respectively summarised in Table 1 and Table 2 for CD and CD-Leontief model fits.

Table A1.1. Pearson correlation coefficients of variables and VIF test for CD fit.

	CD form model fit					
	lnQ	**lnK**	**lnL**	**lnM**	**lnE**	**lnW**
lnQ	1.000	0.031	0.433	0.830	0.829	0.837
InK	0.031	1.000	-0.008	-0.018	-0.003	-0.017
InL	0.433	-0.008	1.000	0.260	0.258	0.267
InM	0.830	-0.018	0.260	1.000	0.993	0.998
InE	0.829	-0.003	0.258	0.993	1.000	0.995
InW	0.837	-0.017	0.267	0.998	0.995	1.000
	Tolerance			**VIF**		
cons-tant	-			-		
lnK	0.977			1.024		
lnL	0.914			1.094		
lnM	0.004			255.025		
lnE	0.009			106.957		
lnW	0.003			353.196		

Table A1.2. Pearson correlation coefficients of variables and VIF test for CD-Leontief fit.

	CD-Leontief fixed model fit			
	LnQ	**lnK**	**lnL**	**InW**
lnQ	1.000	.031	0.433	0.837
InK	0.031	1.000	-0.008	-0.017
InL	0.433	-0.008	1.000	0.267
InM	-	-	-	-
InE	-	-	-	-
InW	0.837	-0.017	0.267	1.000
	Tolerance		**VIF**	
cons-tant	-		-	
lnK	1.000		1.000	
lnL	0.929		1.077	
lnM	-		-	
lnE	-		-	
lnW	0.929		1.077	

Table 3 lists the model fit statistics for both CD and CD-Leontief models. The adjusted R^2 value (0.772) indicates a good model fit in CD functional form results. However the function is not "well behaved" since it does not fulfil the main CD assumption that substitution of all pairs of inputs has a (negative) unit value. In addition, all the parameters, apart from labour, are statistically insignificant. From these regression results, we can conclude that the CD functional form statistically fails and therefore the CD-Leontief Model is more appropriate.

Table A1.3. Model fit statistics and coefficients for CD and CD-Leontief Model.

CD form model fit					**CD-Leontief fixed form model fit**			
	R	**R^2**	**SE****	**Adj.R^2 *****	**R**	**R^2**	**SE**	**Adj. R^2**
	0.869	0.755	0.505	0.722	0.866	0.750	0.497	0.731
	Coefficient	**SE***	**t-test**	**p-test**	**Coefficient**	**SE***	**t-test**	**p-test**
Constant	1.849	6.748	0.27	0.786	6.139	2.043	3.01	0.005
In K	0.074	0.122	0.61	0.548	0.068	0.119	0.56	0.568
In L	0.335	0.131	2.55	0.015	0.349	0.128	2.72	0.010
In M	-0.730	0.947	-0.77	-0.446	-	-	-	-
In E	-0.210	0.590	-0.36	0.724	-	-	-	-
In W	1.484	1.092	1.60	0.182	0.556	0.060	0.37	0.000

*Standard Error of the coefficient estimate, **Standard Error of the model estimate, ***Adjusted R^2

Appendix F: Description of parameters, scalars and variables used in the mathematical model (column 1) and in the model calibration (in column 2) with their relevant units

Indices	Indice	Description	Unit
i	I	Farms	NA
a	A	Agricultural activities	NA
Parameters		**Description**	**Unit**
L_i	Lmax (i)	Total agricultural land on each farm i	Hectare (ha)
p_a	proc (a)	Net profit for each agricultural activity a	Pound per hectare (£/ha)
γ_a	gamma (a)	N residue in soil as a result of agricultural activity a	Kilogram of N per hectare (kg/ha)
$\bar{x}_i$	emiss (i)	Allowed amount of pollution for farm i, calculated by area of farm i and N application norm (50-70 kg/ha multiplied by the available land on farm i)	Kilogram (kg)
$\alpha_{i,a}$	lla (i,a)	Historical land use ratios for each activity a for each farm i	Percentage, unitless
r_{ij}	50/100	Transfer coefficient of N load between farm i and receptor j within one year	Percentage, unitless
Variables		**Description**	**Unit**
$L_{i,a}$	landa (i,a)	Amount of agricultural land allocated to each agricultural activity a on farm i	Hectare (ha)
$\bar{x}_{ia}$	xx (i,a)	Optimal amount of production based on their N output on farm i resulting from agricultural activity a	Kilogram (kg)
q_i	ppp (i)	Amount of pollution allowances bought and sold by farm i	Kilogram (kg)
π	PPPppp	Price of pollution allowances bought and sold by farm i	Pound per kilogram (£/kg)
NA	AuxLand1 (i,a)*	Auxiliary (artificial) variable for land allocated to agricultural activity	Hectare (ha)
NA	AuxLand2 (i,a)*	Auxiliary (artificial) variable for land allocated to agricultural activity	Hectare (ha)
Equations		**Description**	**Unit**
$m_{i,a}$	production (i,a)	Optimal amount of production for each agricultural activity aa on farm i	Kilogram (kg)

e_i	Emissions (i)	Total amount of pollution load as a result of all the agricultural activities on farm i	Kilogram (kg)
$L_{i,a}$	landalloc (i,a)	Lower bound for land allocation to agricultural activity aa based on historical land allocation data (lla)	Percentage, unitless
p_t	tprofit_Tot	Total profit made in the catchment	Pound (£)

* Auxiliary (artificial) variables have no physical meaning, but with their addition, we obtain an initial basic feasible solution for the auxiliary linear programming problem

Appendix G: GAMS code for the optimisation model

The code can be found in the CD attached to the back cover of the thesis.