Developing Engagement Methodologies to Balance Stakeholder Views across Scottish Catchments

Candidate: Kathleen C. Stosch

Thesis submitted for the degree of Doctor of Philosophy
in Environmental Science

Institution: University of Stirling

Supervisors:
Prof David M. Oliver*
Prof Nils Bunnefeld
Prof Richard S. Quilliam

Biological and Environmental Sciences, Faculty of Natural Sciences, University of Stirling, Stirling FK9 4LA, United Kingdom

*Primary supervisor

Submission date: 31/10/2021
Word count: 46 740

Acknowledgements

Firstly, I am immensely grateful to David Oliver, my primary supervisor, for providing me the opportunity, the support and the encouragement to embark on this incredibly rewarding and challenging journey. He has contributed extensively to this thesis through aiding me in critically thinking through research design, methodology and results and by helping to improve the quality of the manuscripts. I also want to thank my other supervisors, Nils Bunnefeld and Richard Quilliam, whose depth and breadth of research expertise and ability to come up with innovative concepts was invaluable in a project focused on developing novel interdisciplinary research methods. I was blessed in having such a practical, supportive and responsive panel behind me.

Special thanks to the Scottish Government Hydro Nation Scholars Programme, for providing funding to support this work and especially to the fantastic folk on the Hydro Nation steering group. Laura Logie, Nikki Dodd, Barry Greig, Bob Ferrier and John Rowan demonstrated a deep and genuine care for the success of each of the Hydro Nation scholars which I appreciate greatly. I would also like to thank my fellow scholars for their camaraderie. Thank you to Mark Tranmer for his comments on the stakeholder network chapter. And thank you to the people at Sawtooth for allowing the use of their software and their services free of charge.

Importantly, I would like to thank my family and particularly my partner Stephen for his vital emotional support and my mum Martina for her support and for proofreading. And thank you to Islay and Niadh for giving me perspective on what is most important in life.

Finally, I would like to thank all the stakeholders, without whom this project would not have been possible, for generously volunteering their time and expertise by participating in the surveys.

Abstract

Ensuring water, food and energy security for a growing world population represents a 21st century catchment management challenge. Failure to recognise the complexity of interactions across socio-ecological systems can risk the loss of key environmental and socio-economic benefits. In particular, the ability of soil and water to meet human needs is undermined by uncertainties around climate change effects, ecosystem service interactions and land use change. Competing stakeholder demands and pressures on land and water resources further complicate catchment management and may lead to land use conflict. Understanding ecosystem service provisioning and trade-offs within catchments, as well as potential synergies and conflict among stakeholder groups, is therefore critical to underpin sustainable use of natural resources. This thesis developed a series of novel engagement methodologies to investigate stakeholder perspectives on opportunities and challenges associated with land and water management in Scottish catchments.

The first objective was to assess trade-offs across catchment uses. Using the production possibility frontier concept, stakeholder assessments of a trade-off between agricultural intensity and the ecological health of freshwater systems were determined and revealed sources of conflict and a diversity of views, especially between environmental regulators and farm advisors. The second objective was to use catchment-scale participatory mapping to identify stakeholder perceptions of land and water management conflicts. This provided spatial detail of the complex combination of land use issues faced by catchment managers. The third objective was to analyse stakeholder networks which identified differences in underlying land and water management issues among the study catchments. The methodology elicited perceived core and periphery stakeholders and those which were not mentioned at all. A final objective was to evaluate the perceived relative effectiveness of agri-environment measures to reduce diffuse pollution on downstream water courses, increase habitat quality to support biodiversity and attenuate flood waters. A best-worst scaling survey of 68 land and water management expert stakeholders revealed "win-win" opportunities for multiple ecosystem service provisioning in Scottish catchments.

The thesis highlights the importance of facilitating increased cooperation and understanding among different stakeholders by quantifying otherwise implicitly held stakeholder views. The methodologies identified potential sources of conflict and likely solutions for win-win opportunities and reinforce the value of accessing and sharing a range of stakeholder perspectives, but also the need to capitalise on this expert knowledge and integrate it into participatory decision-making processes to better manage competing demands on catchment resources.

Table of Contents

5. Characterising stakeholder networks in catchments of contrasting water management issues

6. Expert and non-expert preferences for managing landscapes to deliver multiple benefits from agri-environment measures

List of Figures

List of Tables

1. Introduction

Major economic and technological advancements in the 20th century have lifted millions of people out of poverty and provided water, energy and food to millions more (United Nations Development Programme 2016). As demands of a growing and more affluent global population on natural resources increase, aquatic and terrestrial ecosystems have been degraded beyond repair in many regions, species are becoming extinct at alarmingly high rates, and vulnerability to shocks has increased (Puma 2019; Vörösmarty *et al.* 2010). The new UN development goals continue to focus on lifting people out of abject poverty, while also finding ways to preserve ecosystem functioning. Today still 2.1 billion people are lacking access to safe drinking water and 4.2 billion lack safely managed sanitation facilities (UNESCO 2019); nearly one billion people remain deprived of electricity (IEA (International Energy Agency) 2018); and more than 820 million people have insufficient food (Fears *et al.* 2019). One of the major challenges today is to provide basic human necessities of water, energy and food to all, in an environmentally sustainable, economically viable and socially inclusive manner that is capable to cope with shocks and disasters (Sachs *et al.* 2019). This is relevant for wealthier countries, such as Scotland as well, where due to the interconnectedness of food, water and energy, an increased demand on these resources perhaps may not lead to widespread scarcity, but will present communities with an increasing number of trade-offs, potential conflicts and local issues with resource management (Endo *et al.* 2017).

1.1. Integrated catchment management

The hydrological connectivity that characterises drainage networks at the landscape scale also causes river catchments to integrate multiple stressors from across the landscape (Heathwaite 2010). Abstraction, hydrogeomorphological alterations, and diffuse and point sources of pollution can impede the ability of catchments to supply food, energy and water as well as broader ecosystem services that support human health and well-being. The food-water-energy nexus approach has been developed to manage these three closely interrelated but commonly independently managed resources (Sahle *et al.* 2019). Integrated catchment management approaches further recognise the interconnectedness of catchments and aim to consider the wider conflicts and synergies between management options and the objectives and responsibilities of stakeholder groups (Lerner & Zheng 2011). Recognising catchments as socio-ecological systems that accommodate multiple interests of different stakeholders offers a holistic framework within which to account for the breadth of ecosystem services that catchments host. Both social and ecological systems within a river catchment are complexly intertwined and need to be considered both in research and management and decision-making (Figure 1.1). Realising integrated catchment management is contingent on interdisciplinary science that delivers novel, cutting-edge outputs, which allow the cooperation of multiple fields of research, stakeholder groups and quantify trade-offs between management options within the water-energy-food nexus, while considering wider ecosystem service provisioning. There is a need, therefore, to quantify and optimise the range of multiple benefits that catchments can deliver in response to shifts in their management and the wider environment.

1.2. Interdisciplinary participatory research

Conducting interdisciplinary participatory research with land and water management stakeholders may not only help identify differing views between stakeholder typologies (Darvill & Lindo 2016), but also facilitate effective knowledge exchange among stakeholders, while also capitalising on important expertise and understanding which may be otherwise missed (Galafassi *et al.* 2017). Participatory mapping techniques can aid understanding of the spatial distribution of conflicts and social benefits, especially for cultural ecosystem services, which are difficult to estimate (Reilly *et al*. 2018; Brown *et al.* 2020). The use of participatory approaches are therefore vital for integrating the social demands on land and water management, which are often neglected, and hence may avoid potential conflict of natural resource use and management (García-Nieto *et al.* 2015). Participatory research approaches can also increase the inclusiveness and social acceptability of management decisions, and ensure they are implemented more effectively (Etienne *et al.* 2011; Oliver *et al.* 2017). To date, relatively few participatory mapping studies have used both quantitative and qualitative

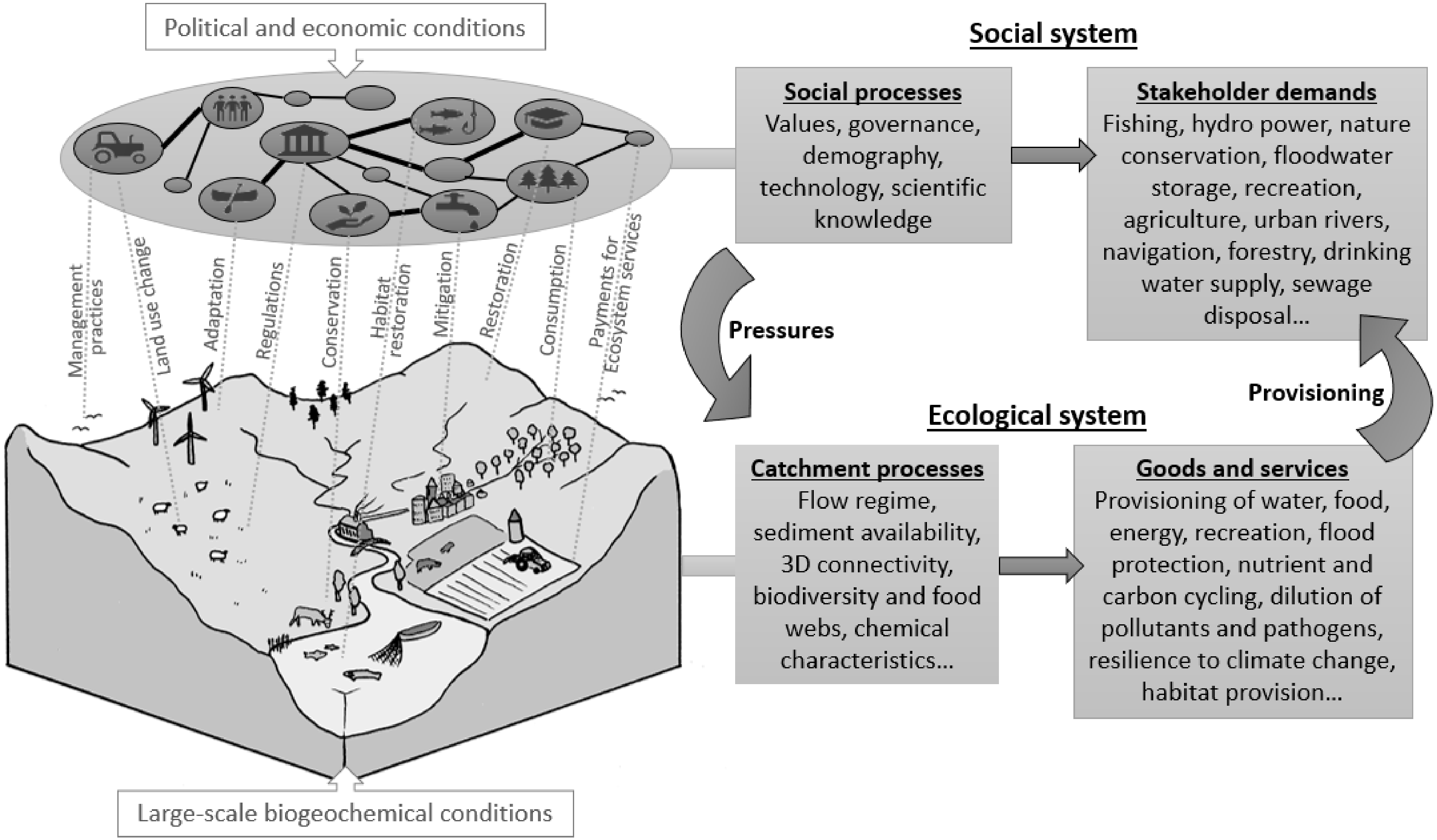


Figure 1.1: Illustration and conceptual model of the social and ecological system in a river catchment and the interactions between them.

data to complement and validate their methodologies (Brown *et al.* 2017), and there is a dearth in general of empirical studies on stakeholder engagement using quantitative methods (Kujala *et al.* 2022). Hence there is a need for interdisciplinary research outputs that are catchment-centric and quantify multiple stakeholder views and multiple ecosystem service benefits in land and water management.

Throughout the following chapters we use the definition of stakeholders as those individuals, groups, or organizations that influence or are affected by catchment management (Freeman 2015), and define stakeholder engagement as actively soliciting the knowledge, experience, judgment and values of individuals selected to represent a broad range of direct interests to particular issue (Deverka *et al.* 2012). Stakeholder engagement from a catchment management perspective has the benefits of capturing knowledge, increasing ownership of projects, reducing conflict, encouraging innovation and facilitating partnerships; however, there is also an ethical dimension, where meaningful stakeholder engagement may enhance inclusive decision making, promote equity, enhance local decision making and build social capital (Mathur *et al.* 2008). Stakeholders incorporate their knowledge into socio-ecological systems, but also their values and beliefs, and their assumptions and biases (figure 1.1). Hence it is important acknowledge the political dimension of stakeholder engagement and representation and be cautious of reinforcing inherent power relationships both in research and in catchment management.

Here we focused on methodological development of stakeholder engagement tools and engaged primarily with key land and water management stakeholders to aid proof of concept of the methodologies. It may be noted that these key stakeholders, already have power in the planning and decision-making process and future use of the methods developed here may be extended to help empower more local actors such as local grassroots organisations and local citizens (Thaler & Levin-Keitel 2016).

1.3. Aims and objectives

The overarching aim of this project was to quantify stakeholder views on land and water management in Scottish catchments and deliver strategies to promote stakeholder collaboration and ameliorate conflict. This was achieved by developing novel, mixed method, stakeholder engagement methodologies to elicit views from key land and water management experts. The objectives were to:

(1) Assess the trade-offs and synergies between catchment uses and find ways to optimise landscape scale ecosystem service provision in Scottish catchments.

(2) Quantify how stakeholder views differ among key groups of stakeholders and across different catchments with diverse and contrasting geomorphologies, land cover types, stakeholder communities, and land and water management pressures.

(3) Develop methods that can reduce stakeholder conflict by facilitating cooperation and building shared mutual understanding.

(4) Determine the practical relevance of the participatory methodologies for land and water management planning and decision-making.

1.4. Thesis structure

The thesis is organised as six chapters following this introduction. Chapter 2 is a critical review outlining research priorities. Chapters 3-6 document the major scientific findings, and chapter 7 provides a synthesis of the research findings and identifies opportunities for future research. The critical review and first data chapter are published in peer-reviewed journals, listed in Table 1.1, and appear as they do in print but without their abstracts.

Table 1.1. Details of published chapters.

Chapter	Details
2	**Stosch, K. C.**, Quilliam, R. S., Bunnefeld, N., & Oliver, D. M. (2017). Managing multiple catchment demands for sustainable water use and ecosystem service provision. *Water*, *9*(9), 677. DOI: https://doi.org/10.1016/j.scitotenv.2018.10.090
3	**Stosch, K. C.**, Quilliam, R. S., Bunnefeld, N., & Oliver, D. M. (2019). Quantifying stakeholder understanding of an ecosystem service trade-off. *Science of the Total Environment*, *651*, 2524-2534. DOI: https://doi.org/10.1016/j.scitotenv.2018.10.090

2. Managing Multiple Catchment Demands for Sustainable Water Use and Ecosystem Service Provision

Published in Water 2017

Manuscript details:

Stosch, K. C., Quilliam, R. S., Bunnefeld, N., & Oliver, D. M. (2017). Managing multiple catchment demands for sustainable water use and ecosystem service provision. *Water*, *9*(9), 677.

DOI: https://doi.org/10.1016/j.scitotenv.2018.10.090

Author contributions:

K. C. Stosch conceived the rationale, carried out the literature review, and wrote the manuscript. D. M. Oliver, R. S. Quilliam and N. Bunnefeld provided advice on the literature review process and added critique and suggestions to improve the review at each draft stage. Anonymous reviewers suggested improvements to the final manuscript.

2.1. Introduction

Catchments have been widely altered through large-scale land cover and land use change, including industrialisation, urbanisation, intensive agriculture and via hard-engineering, i.e., the construction of infrastructure designed to divert water for drinking, irrigation or hydropower schemes (Sterling *et al.* 2012). This has often benefitted economic productivity but has also frequently led to unintended consequences, such as reduced water quality and ecosystem functioning, and reduced resilience against other pressures such as invasive species and climate change (Ormerod *et al.* 2010). Freshwater environments provide vital benefits to humans, and scientists and policy-makers increasingly look to ecosystem service theory and assessments to support sustainable catchment management. Ecosystem services are defined as the broad range of goods and services that an ecosystem provides, which enhance human health and well-being (Millennium Ecosystem Assessment (MEA) 2005). However, many of these services may be threatened where the upstream catchment system, within which a given waterbody is located, is poorly or inappropriately managed (Green *et al.* 2015).

The recognition of catchments as a socio-ecological continuum that accommodates multiple interests of different stakeholders therefore offers a holistic framework within which to account for the breadth of ecosystem services that catchments host (Likens *et al.* 2009). By assessing stakeholder interests across catchment functioning, from headwaters through to estuaries and bathing zones, we can begin to account for negative (trade-off) and positive (synergistic) interactions between ecosystem services that may arise from competing stakeholder interests (Bennett *et al.* 2009). Interactions and interdependencies between ecosystem services present a principal challenge for catchment management as it makes it difficult to predict the outcome of planning or mitigation options designed to tackle a particular environmental issue. Understanding such interactions is further complicated due to the spatial and temporal variability around natural processes, the occurrence of thresholds and 'tipping-points' in environmental systems, non-linearity and the potential for irreversible collapse of services in catchments (Spears *et al.* 2012; Tromp-Van Meerveld & McDonnell 2006).

The interconnectedness of catchment attributes has long been recognised, and underpins the concept of integrated catchment management (ICM; Lerner & Zheng 2011). ICM, similarly to integrated water resource management, is a holistic approach which considers the wider conflicts and synergies between management options and the objectives and responsibilities of stakeholder groups (Lerner *et al.* 2010). With unprecedented shifts in climate and land use change, and a rapidly changing political climate, it is essential that ICM evolves to balance competing demands for different ecosystem services across what has been coined the water-energy-food (WEF) nexus. WEF nexus thinking is an approach for integrated natural resource use which considers the interconnectivity between human resource use and the challenges of providing water, food and energy security for a growing global population (Biggs *et al.* 2015). WEF science represents an opportunity for truly interdisciplinary working and transformative research, which can benefit ICM and the ecosystem services approach to environmental management given the fertile, conceptual space that overlaps the disciplines of catchment science and ecological economics, ecological politics, remote sensing, and even computer game research. Arguably, the paradigm of ICM has become complacent with its acknowledgement of the need for interdisciplinary science rather than being fully exploited to deliver novel, cutting-edge interdisciplinary frameworks, i.e., practical outputs, to promote transformative catchment management, which allows the cooperation of multiple fields of research, stakeholder groups and quantifies trade-offs between management options within the WEF nexus, while considering wider ecosystem service provisioning. Thus, a more serious interdisciplinary research ambition needs to replace the superficial transdisciplinary rhetoric, such as at the economics and ecology interface (Spash 2012). There is a need, therefore, to quantify and optimise the range of multiple benefits that catchments can deliver in response to shifts in their management and the wider environment. Subsequently, the aims of this review are to: (i) define and critically review ecosystem trade-off research from a catchment perspective (Section 2); (ii) critically evaluate the benefits and challenges of utilising ecosystem service-based approaches for catchment management (Section 3); and (iii) propose a roadmap of future research to account for ecosystem trade-offs and co-benefits in decision-making and in turn promote more integrated catchment management (Section 4), supported by trans-disciplinary science. An extensive range of source material was identified, analysed, synthesised and evaluated for this review. The identification was conducted initially through a comprehensive web search of the major relevant concepts and topics using Web of Science and Google Scholar. A subsample of these contributions was selected based on the relevance of their titles initially, and their abstracts subsequently. Further contributions were selected by using a snowballing approach, which identified papers cited in reference lists but also determined subsequent publications that cited existing sources. The exception to this is Figure 2 which is based on the first author's data collection from an engagement exercise which captured the perception of 18 stakeholders (academics, environmental regulator staff, NGO staff and farmer's union staff) on ecosystem service provisioning in Scottish upland and lowland catchments at a research conference in Edinburgh in 2016.

2.2. Ecosystem Service Trade-Offs and Co-Benefits from a Catchment Perspective

Patterns of hydrological connectivity that generate the characteristic drainage network at the landscape scale are also responsible for the integration of water quality signatures from multiple stressors (Heathwaite 2010), for example, abstraction, hydrogeomorphological alterations, and diffuse and point sources of pollution (Elosegi & Sabater 2013). Climate change impacts, and pressures to secure food, energy and water provision for nine billion people by 2050, combine to elevate and accelerate the magnitude of effects from such stressors in catchment systems. Understanding how different parts within the WEF nexus interact and how management decisions may affect ecosystem functioning and service provisioning is vital to manage catchments sustainably. This section defines and critically reviews ecosystem trade-off research from a catchment management perspective.

2.2.1. The Ecosystem Service Concept and Catchment Management

The concept of ecosystem services is useful for catchment management because it provides a framework within which different functions and services of catchment systems can be recognised as part of a larger complex and interlinked landscape, and one where management options of one service may impact, positively or negatively, upon other services. Using the ecosystem service concept may hence aid cross-sectoral interaction and collaboration by highlighting the linkages between catchment management and ecosystem service provisioning, and by providing a common reference language to help facilitate cooperation among stakeholder groups and research disciplines (Galler *et al.* 2016).

As part of the UN Millennium Ecosystem Assessment (Millennium Ecosystem Assessment (MEA) 2005), ecosystem services were classified into three broad categories: provisioning (producing resources), regulatory (regulating processes in the natural environment), and culturally beneficial services. Supporting services sustain these three categories, and may be more appropriately named ‘underlying ecosystem processes’, as they do not directly benefit humans. While the ecosystem service concept has been criticised for being anthropocentric and capitalist (Gomez-Baggethun & Ruiz-Perez 2011), there are clear benefits of quantifying the services that the natural environment can provide, which would otherwise be undervalued. As a result, recent research has begun to map the spatial provision of ecosystem services across landscapes (de Araujo Barbosa *et al.* 2015; Landuyt *et al.* 2014; Martínez-Harms & Balvanera 2012) and a number of approaches have taken the ecosystem services concept further in an effort to improve conservation and restoration projects and underpin other environmental decision-making (Fu *et al.* 2015; Trabucchi *et al.* 2012; von Stackelberg 2013), assess possible synergies and trade-offs between services (Bennett *et al.* 2009; Butler *et al.* 2013; Howe *et al.* 2014), or evaluate payment options for ecosystem services (Derissen & Latacz-Lohmann 2013; Matthies *et al.* 2016). Multi-criteria modelling frameworks have also been developed to further provide decision support on sustainable ecosystem service provision (Bohnet *et al.* 2011; Wam *et al.* 2016). These can allow stakeholders to evaluate the outcomes of changes in land management and how they may affect wider ecosystem services.

2.2.2. Trade-Offs between Ecosystem Services

How people manage ecosystems for certain services will impact on the type, magnitude and relative composition of wider ecosystem service provision across catchment landscapes (Cordingley *et al.* 2016; Rodríguez *et al.* 2006). The interdependency of services presents a principal challenge for ecosystem and thus catchment management. Pair-wise interactions between ecosystem services can be thought of as either 'trade-off' or 'win-win' scenarios (Table 2.1); however, most ecosystem service interactions involve multiple provisioning, regulating and cultural ecosystem services. Interactions are further complicated as they usually involve many 'bundles' of ecosystem services which makes them highly difficult to predict. Bundles are groups of co-occurring, interacting ecosystem services that are provided from a certain area in the landscape (Raudsepp-Hearne *et al.* 2010). A woodland, for instance, may provide timber and wild foods for foraging and hunting, carbon sequestration, pollutant and flood buffering, and recreational benefits (Wam *et al.* 2016). When the woodland is degraded or its area is reduced, the entire bundle of services would also be reduced. The detailed illustrations and descriptions of ecosystem services in a European temperate grassland by Pilgrim *et al.* (2010) show how complex ecosystem service interactions can be, even in a relatively simple and well-studied ecosystem.

Table 2.1: Examples of pair-wise trade-offs (negative interaction) and win-win (positive interaction) relationships between provisioning (P) and regulating (R) ecosystem services. Impacts may be expressed either locally, downstream, or in the wider environment.

Driver	**Service A**	**Service B**	**Scenario**	**Spatial Scale**	**Reference**
Fertiliser use	crop production (P)	water quality (R)	trade-off	downstream	(Ewing & Runck 2015)
Forest harvesting	timber production (P)	runoff, water quality (R)	trade-off	downstream	(Costa *et al.* 2003)
Afforestation	carbon sequestration (R)	water quantity (P)	trade-off	downstream	(Engel *et al.* 2005)
Crop irrigation	crop production (P)	soil salinisation (R)	trade-off	local	(Dehaan & Taylor 2002)
Diffuse pollution buffer areas	water quality (R)	soil, air & groundwater quality (R)	trade-off	local, downstream & wider environment	(Stevens *et al.* 2009)
Constructed wetland	water quality (R)	biodiversity (R)	win-win	local & downstream	(Semeraro *et al.* 2015)
Wetland restoration	water quality (R)	fisheries (P)	win-win	downstream	(Butler *et al.* 2013)
Habitat protection	pollination (R)	crop production (P)	win-win	local	(Kasina *et al.* 2009)
Lake restoration	water quality (R)	human health (R)	win-win	downstream	(Carvalho *et al.* 2013)
Farmland forest	pest control (R)	coffee production (P)	win-win	local	(Karp *et al.* 2013)

Trade-offs occur most often when an ecosystem is managed to increase or maintain a single service which causes the reduction of other services, or if services react to a common driver, such as land use change or climate change (Bennett *et al.* 2009). Humans have generally altered natural ecosystems (Figure 2.1a) to provide greater provisioning services, such as food production (Figure 2.1b); however, intensive management of catchments often significantly reduces regulating and cultural services which are not directly valued, such as carbon sequestration, river biodiversity, lake amenity values and human health (Raudsepp-Hearne *et al.* 2010).

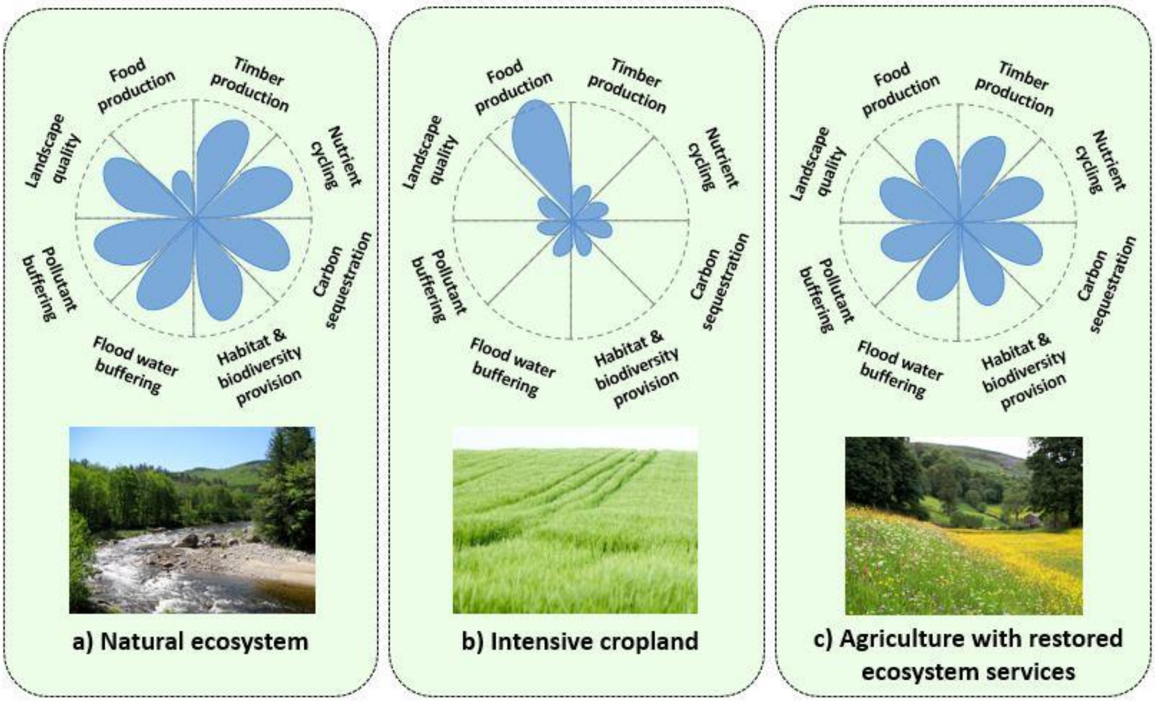


Figure 2.1: 'Flower diagrams' illustrating multiple ecosystem service provision under three hypothetical land uses, which are (**a**) natural; (**b**) intensively farmed and (**c**) managed for multiple ecosystem service provision. Ecosystem service provision is indicated along the axes, which are not labelled or normalised for this qualitative illustration. Adapted with permission from Foley *et al.* (2005).

There are also examples of trade-offs between two regulating services, such as due to pollution swapping. Capturing diffuse pollutants in sedimentation ponds or buffer strips may reduce water pollution, but can increase soil, groundwater and air pollution ((Stevens *et al.* 2009); Table 2.1). Environmental externalities may compromise ecosystem functioning to such an extent that they compromise the targeted provisioning service itself. For example, irrigation-induced soil salinisation in the Murray-Darling Basin is estimated to cost Australia US$200 million annually in lost agricultural production (Dehaan & Taylor 2002).

Trade-offs may be the result of an explicit management choice, but are often unintentional due to a lack of knowledge about ecosystem service interactions and the technical expertise to make decisions that benefit more than a single service. Even if a trade-off is intentional, there may be unwanted effects at different scales to those considered, especially at larger spatial and temporal scales (Rodríguez *et al.* 2006). Ecosystem service trade-offs may be classed along three axes depending on spatial scale, temporal scale and how reversible they are (Rodríguez *et al.* 2006).

Temporal short-sightedness may be either predetermined or inadvertent, for example, as part of a political system, which works in short timescales of four to five years. An example of temporal externalities is the extraction of groundwater beyond replenishment rate, which may not yet significantly impact the current generation. Temporal trade-offs may also occur due to system lags and delays caused by hydrological and biogeochemical processes, meaning rivers may take decades to respond to reduced nutrient inputs to agricultural land (Hamilton 2012; Jarvie *et al.* 2013) and continue to be affected by so-called 'legacy' concentrations of pollutants, such as phosphorus. In some trade-offs, ecosystem function can be changed in such a way that it leads to regime shifts which may not be reversible (Gordon *et al.* 2008), such as when a lake shifts to a eutrophic state, altering its chemical and biological makeup.

Ecosystem service trade-offs may be expressed at a local to a catchment or even global scale and ecosystem services are valued differently by stakeholders depending on the scale of interest (Hein *et al.* 2006). An individual farmer's land is most profitable if it is entirely cultivated, however, riparian buffer strips may have a greater non-monetary ecosystem service provision on a catchment scale, which is why governments pay farm subsidies to allow some land to be taken out of production. Spatial trade-offs in catchments are often expressed downstream which means upstream users are less likely to experience negative effects, which then drives the potential for conflict between upstream and downstream users (Asquith *et al.* 2008). For example, there is a clear trade-off between the benefit of abstracting water upstream for a particular human use against the dis-benefits of reduced flows downstream, which may cause conflict between different communities or entire nations (Munia *et al.* 2016). Some effects may be attenuated downstream due to pollutant and flood water buffering, however, if nutrient export and drainage are increased in the upper catchment, in-stream pollution and flooding are likely to increase for stakeholders in the lower catchment. Generally, upland catchments provide many key regulating ecosystem services, as lowland catchments are often more suited for agricultural production and human settlements (Figure 2.2).

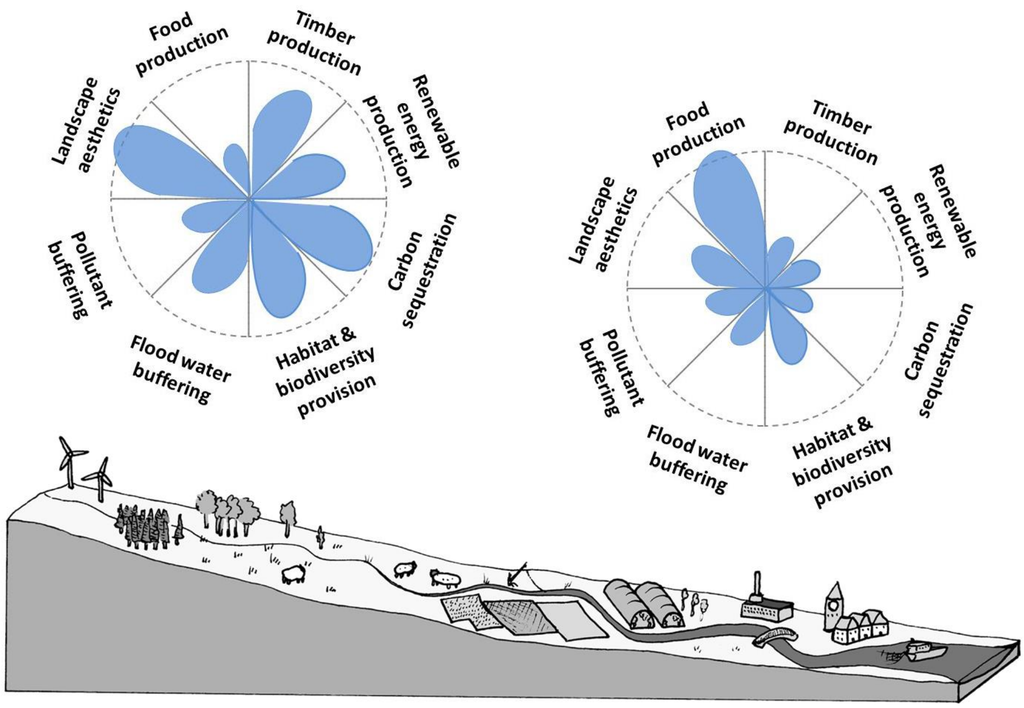


Figure 2.2: 'Flower diagrams' illustrating perceived multiple ecosystem service provision in an upper and lower catchment. Ecosystem service provision was indicated by stakeholders (academics (*n*=9), environmental regulator staff (*n*=5), NGO staff (*n*=3) and farmer's union staff (*n*=2)) to estimate how they perceive provisioning in a generic Scottish upland and lowland catchment. The stakeholders were engaged during a conference on sustainable management of Scottish farms in spring 2016, and asked to draw the petals of the flower diagram with a white board marker on a laminated A3 print of two unfilled petal diagrams. The drawings were then digitalised to allow counting of rasters in each petal. Mean (± SE) perceived ecosystem service provisioning (as a percentage of the total provisioning of the services in both parts of the catchment (total = 200%) for the upper (U) and lower (L) catchment were: Food production: U = 6.8 ± 0.9, L = 28.1 ± 3.0; Renewable energy production: U = 12.6 ± 1.7, L = 6.3 ± 1.3; Timber production: U = 19.5 ± 1.9, L = 7.0 ± 1.4; Carbon sequestration: U = 24.8 ± 3.0, L = 5.7 ± 0.8; Habitat & biodiversity provision: U = 22.8 ± 1.9, L = 10.1 ± 1.9; Flood water buffering: U = 17.8 ± 1.9, L = 8.0 ± 1.3; Pollutant buffering: U = 8.7 ± 1.8, L = 7.2 ± 1.2; Landscape aesthetics: U = 28.8 ± 2.0, L = 10.8 ± 1.6.

2.2.3. Win-Win Scenarios and Managing Catchments for Multiple Ecosystem Service Provision

In contrast to trade-offs, 'win-win' situations occur when positively correlated services are enhanced concurrently through explicit management interventions. Land-based management options that limit nutrient loss from agricultural land, for instance, may save money for the farmer while also improving in-stream water quality, which in turn may benefit aquatic ecosystems and public health. Other management techniques, such as agricultural diversification and environmentally focused land management plans may benefit aquatic biodiversity without compromising farm incomes (Stoeckl *et al.* 2015). Trade-offs in ecosystem service are, however, more common than these synergistic relationships due to competing social, economic or ecological goals (Howe *et al.* 2014). Accounting for these competing factors may, however, allow interventions to be targeted at increasing the likelihood of win-win situations and the meeting of multiple demands.

Managing catchments for multiple ecosystem service provision may reduce the output of provisioning services significantly, but increase ecosystem functioning and human health and well-being overall (Figure 1c; (Foley *et al.* 2005)). Confronting the WEF challenges of the future, while preserving regulating and cultural ecosystem services, will require integrated management of those trade-offs driven by catchment governance, and a strategic move towards improved ecosystem service provision will be essential in order to underpin sustainable catchment management.

2.2.4. Multiple Stakeholder Preferences within a Trade-Off

Trade-offs may occur due to biophysical constraints within an ecosystem, but conflicts may also arise due to diverging preferences held by stakeholders (Martin-Lopez *et al.* 2012). The ecosystem service concept is socio-ecological and therefore not only depends upon the biogeophysical constraints of an ecosystem, but also on how people value the benefits and services that an ecosystem provides. When aiming to quantify and optimise ecosystem service provisioning in catchments one needs to be aware that different management scenarios may be not acceptable for people in certain parts of a catchment, or for certain stakeholder groups. Cavender-Bares *et al*. (2015) developed an approach to integrate the biophysical and social trade-off between two services by balancing the preferences of stakeholders (illustrated in Figure 2.3). The production possibility frontier (PPF), shown as the black line in Figure 2.3, represents the balance between agricultural yield and downstream water quality in this example (Figure 2.3a). The shape of the PPF depends on the biogeophysical constraints of an ecosystem and can be changed through management practices and technology, for example, a catchment with very deep soils may be able to buffer excessive nutrient input, and therefore retain high water quality, in turn increasing crop yields. The curve may be moved upwards via the implementation of management options that increase both water quality and yield, e.g., through efficient fertiliser use, buffer zones or intercropping (Ewing & Runck 2015).

Isoclines of stakeholder utility values may also be plotted over the graph to model the preferences of stakeholder groups, with darker lines representing greater utility (Figure 2.3b). The point at which these meet the PPF represents the maximum sustainable utility value that a stakeholder may attain under these certain biophysical constraints of the PPF. Using these functions, the trade-off preferences of multiple stakeholders may be plotted, which shows the potential conflict between their positions (Figure 2.3c). This approach can thus reveal potential stakeholder conflict as each trajectory concomitantly increases utility for one stakeholder group, whilst decreasing it for the other. This subsequently provides insight into how valuable services are to different stakeholder groups, but more importantly also revealing potential sources of conflict and synergies among stakeholder groups. Using PPF and utility functions may reveal opportunities for win-win outcomes or identify whether stakeholders could be offered compensatory payments for utility losses (King *et al.* 2015). This concept could also be used in catchment modelling to find minimum or optimum levels for certain ecosystem service provisioning or may simply be used as part of a participatory approach to engage stakeholders, allow discussion on barriers and conflicting preferences, and build shared mutual understanding to facilitate future cooperation.

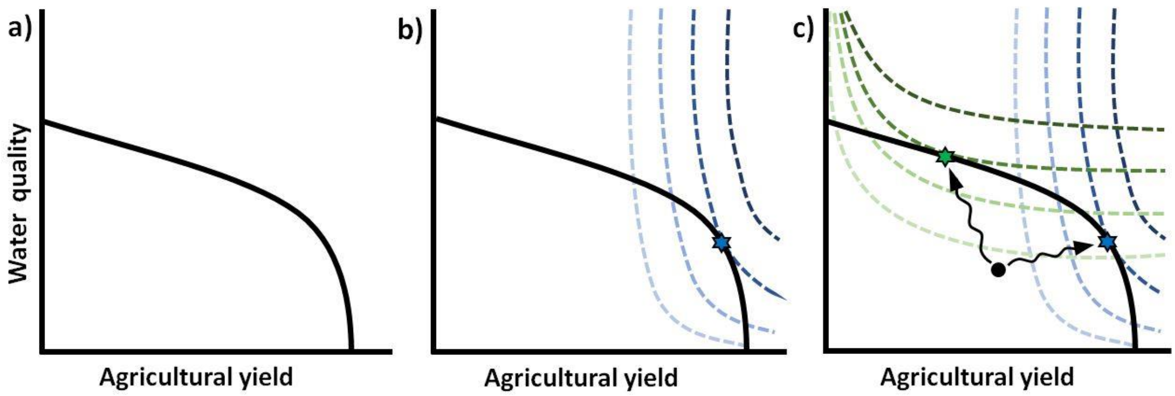


Figure 2.3: Illustrating the trade-off between water quality and agricultural yield using a production possibility frontier (PPF; black line, **(a)**), and the utility functions of two stakeholders with differing values (shades of green or blue dotted lines **(b)**). The blue star illustrates the ideal balance of the trade-off for upstream farmers, whereas the green star shows the preferences of downstream stakeholders, who are impacted by lowered water quality, such as people who use or manage downstream bathing waters **(c)**. Reproduced from King *et al.* (2015).

2.3. Using the Ecosystem Service Concept for Catchment Management

In the 20th century, water managers relied on hard engineering solutions such as dams and centralised water treatment plants to meet human demands for water. As it became more apparent that past management of freshwater had degraded ecosystem functioning, there was increased realisation of alternative soft-path solutions (Gleick 2003; Vörösmarty *et al*. 2010). Policy-makers and water companies increasingly recognised that catchment-based solutions, such as wetland restoration, may not only deliver multiple benefits for water quality improvements, flood buffering, carbon storage and habitat provision, but may be more cost-effective than engineering structures (Costanza *et al.* 2014; Ervin *et al.* 2012; Smith & Porter 2010). The following will critically review the benefits and challenges of utilising ecosystem service-based approaches for catchment management.

2.3.1. Ecosystem Service-Based Approaches and Integrated Catchment Management

ICM recognises that to manage natural resources sustainably we need to understand how the different socio-ecological components of catchment ecosystems function and interact. The underpinning philosophy of the ICM process is to identify potential synergies and conflicts of different management options within an entire catchment and assess how they could remediate existing problems or those that may arise in the future. Integrated ecosystem-based management, has so far been more widely utilised in marine ecosystem management (Levin *et al.* 2013; Pendleton *et al.* 2015; Vasslides & Jensen 2016), but has potential for catchment management due to the strong interactions between ecosystem services and the involvement of multiple stakeholders. These so-called ecosystem service-based management approaches should assess the delivery of ecosystem services and disservices, and aim to better understand ecosystem functioning and interdependencies while acknowledging uncertainties (Martin-Ortega *et al.* 2015). To achieve this, natural and social sciences need to be integrated to co-produce stakeholder-driven decision-making tools, which are both socio-ecologically sound and valuable to decision-makers, and align with emerging agendas in WEF nexus science. Stakeholder mapping tools can help identify the groups which may influence decision-making and those who may be impacted by it (Sova *et al.* 2016). This approach allows water managers and decision-makers to consider how an entire socio-ecological system can function (including intangible services and less influential stakeholder groups), which can facilitate integrated catchment management, and in the long-term can help improve catchment functioning and service provision. However, while the value of adopting ICM and ecosystem service-based approaches is clear, their implementation can be challenging; they require significant resources to allow robust ecosystem service assessment and necessitate integration of stakeholder groups into the decision-making process along with representation of, and respect for, both natural and social perspectives.

2.3.2. Ecosystem Service Valuation Methods

The ecological value of aquatic systems may be measured by indicators of its health state, such as biodiversity, water quality or combined indicators such as the EU Water Framework Directive's (WFD) ecological status (Martinez-Haro *et al.* 2015). Socio-economic values can be categorised as either 'use values' from consumptive goods, such as crops or timber, or non-consumptive 'non-use values', which can either be direct, such as landscape aesthetics or recreation, or indirect, such as nutrient cycling, erosion control or floodwater buffering (de Groot *et al.* 2010). Economists may estimate non-market values based on the cost of alternatives, such as installing a water treatment plant to replace natural water-purification services, or the cost of flooding to properties if a wetland has been removed (Keeler *et al.* 2015; Liquete *et al.* 2012; Liu *et al.* 2010; Martin-Lopez *et al.* 2012). For many ecosystem services, however, these revealed preference methods are not possible and valuation has to rely on stated preference from surveys, which makes it inherently difficult to accurately estimate non-market values, and there remain key gaps in knowledge around the value of ecosystem services. This is particularly true for cultural services, provisional services from genetic and medicinal resources, and regulating services such as seed dispersal and resistance to pests, pathogens and invasive species (Harrison *et al.* 2010). Furthermore, ecosystem service function may supply whole bundles of services that are often overlooked due to difficulties in their valuation. To value water quality related ecosystem services, for instance, a range of services need to be considered, such as drinking water treatment costs, human health benefits, and recreational opportunities (Keeler *et al.* 2012). There are also limitations to how accurately ecosystem services can be assessed, i.e., due to natural variability of services through time and space, and our lack of understanding of how ecosystem functioning supports ecosystem service provision (Hou *et al.* 2013). Yet, exact valuations may not always be necessary, for example, if determining a management option with the greatest benefits for multiple ecosystem service provision (Gilvear *et al.* 2013).

The anthropocentric nature of the ecosystem service concept has led to a critique of ecosystem service valuation with suggestions that it may promote the 'commodification' of nature and lead to the degradation of parts of ecosystems which are not valued within ecosystem service assessments (Kosoy & Corbera 2010). To counter that critique it is important to highlight that a number of natural resources are already valued via economic markets and so by valuing multiple ecosystem services it becomes possible to extend valuation beyond a few tangible services to those catchment resources that provide wider ecosystem functioning and human health benefits. Expressing the value of ecosystem services in monetary terms does help to raise awareness of the importance of ecosystems and biodiversity amongst the public and politicians, and enables more cost-efficient targeting of limited funds for protection and restoration (de Groot *et al.* 2012). It is necessary, however, to remain aware of the limitations of ecosystem service valuation if they are being used as part of ICM due to the difficulty of accurately valuing services. Even services which can be expressed in monetary terms may not be directly compared to other services and may be valued differently by particular stakeholders. If there is a disparity between service valuation of upstream and downstream catchment users payment systems may be set up to balance ecosystem service utility.

2.3.3. Payment for Ecosystem Services (PES)

The majority of ecosystem services from catchments, e.g. soil quality, may be classed as societal and cannot be paid for by the end-user. Many EU governments therefore intervene to stop the erosion of these services by investing in their protection, such as through agri-environment schemes (Honey-Rosés *et al.* 2013). A critique of current EU payment schemes is that they are not particularly well targeted to improve soil and water quality in catchments, but focus mainly on biodiversity and carbon sequestration (Pe'er *et al.* 2014). Utilising user-financed 'payment for ecosystem services' (PES) schemes, where possible, can improve cost-effectiveness of catchment-scale management and can be locally targeted and monitored combining water quality and biodiversity (Ortega-Pacheco *et al.* 2009; Wunder *et al.* 2008). Although PES is not a panacea that can address all environmental issues, it provides a useful tool that can be tailored to avoid the erosion of non-market ecosystem services and services with challenging spatial and temporal scales, and balance competing demands of upstream and downstream catchment users (Engel *et al.* 2008). PES have potential to benefit ICM by incentivising management options that benefit the common good and improving downstream ecosystem services such as by buffering pollutants or flood water and helping decision-makers to recognise the value of the loss of wider ecosystem services catchments provide (Bellver-Domingo, A. Hernández-Sancho, F. Molinos-Senante 2016; Hack 2015). Another option for making PES schemes more effective is the development of spatially targeted decision support tools that take into account ecological, financial and social constraints of different management options and how they may impact on multiple ecosystem service provision (Uthes & Matzdorf 2013).

2.3.4. Ecosystem Service Assessment, Modelling and Mapping

Over the past decade, research on ecosystem service assessment and mapping has increased substantially (Nelson & Daily 2010; Seppelt *et al.* 2012). Due to their size and the spatial and temporal heterogeneity of ecosystems, tools such as geographic information systems (GIS) are regularly used for visualising and analysing landscape ecosystem service provision (Nemec & Raudsepp-Hearne 2013). The scale and rationale for ecosystem service mapping studies varies greatly from large-scale global and continental studies, to catchment-scale investigations, identifying broader patterns of spatial ecosystem service distribution or researching more in-depth trade-off analyses, changes in ecosystem services, or the prioritisation of areas for planning and management (Egoh *et al.* 2009; Harrison *et al.* 2010; Raudsepp-Hearne *et al.* 2010).

There are a growing number of approaches for ecosystem service mapping, e.g., spatial mapping can be informed via modelled outputs such as modelled nutrient runoff, or from direct measurements and observations, such as water quality (Maes *et al.* 2012; Nemec & Raudsepp-Hearne 2013). Process-based models can offer a more dynamic (rather than static) assessment of ecosystem services under changing ecosystem variables and therefore can provide valuable decision support (Nemec & Raudsepp-Hearne 2013). Value transfer methodologies, which assign a total economic value to certain land cover types using GIS are also widely used for assessing ecosystem service provision (i.e. (Costanza *et al.* 1997)). Different modelling tools may be applied for ecosystem service assessments in ICM, such as InVEST (Integrated Valuation of Ecosystem Service and Tradeoffs) mapping to estimate water yield and consumption for croplands in California (Matios & Burney 2017), or the SWAT (Soil and Water Assessment Tool) to estimate stream flow, sediment yield, or surface runoff in catchments (Francesconi *et al.* 2016). Ecosystem service mapping can be a powerful tool for ICM as it illustrates spatial trade-offs within a catchment and may help highlight potential synergies or conflicts of different management options.

2.3.4.1. Challenges and Limitations of Ecosystem Service Assessments

To make well-informed decisions about trade-offs between different management options, it is necessary to assess multiple ecosystem services at multiple scales, and to improve our knowledge of ecosystem service provision and valuation (de Groot *et al.* 2010). Most ecosystem service studies, however, focus on a small selection of services, and the more difficult to estimate cultural services are frequently omitted (Martínez-Harms & Balvanera 2012). A large proportion of mapping studies lack sufficient scale-appropriate data, use secondary data more frequently than raw data and do not validate their modelled results (Martínez-Harms & Balvanera 2012). Seppelt *et al*. *(2011)* concluded that less than a third of reviewed ecosystem service studies provided conclusions that were soundly based on science; and found that many studies lacked primary data, validation or quantitative assessment of uncertainties. Less than 40% of the reviewed ecosystem service studies make conclusions based on primary measurements or observations, most likely due to the expense and difficulty of collecting primary data (Seppelt *et al.* 2011). Furthermore, most ecosystem service models fail to account for basic ecological concepts such as species interactions or Island Biogeography Theory (Maes *et al.* 2012). Ecosystem services not only correspond to the ecological functioning of landscapes, but also to stakeholders' socio-economic and cultural value systems (Martin-Lopez *et al.* 2012). Due to these limitations of ecosystem service assessments to holistically capture the benefits that catchments provide, they may not be used to solely inform sustainable decision-making without the input of experts and local stakeholders.

Both environmental and socio-economic systems are complex and variable in space and time, which makes it inherently difficult to accurately model and map ecosystem services in catchments. There are uncertainties that arise from the inherent variability of the stochastic and often chaotic nature of natural phenomena, such as extreme weather events, and also from societal variability due to socio-economic and cultural dynamics, which can include chaotic and unpredictable drivers, e.g., wars and technological developments (Walker *et al.* 2003). In addition, structural uncertainty in model design can arise due to a lack of understanding of the true biophysical processes that govern some complex environmental systems. When assessing the provision of ecosystem services, it is particularly important to be explicit about the uncertainties linked to our limited understanding of how ecosystem services are generated and how they may interact, particularly as many ecological quality indicators respond non-linearly to underlying pressures, and may display multiple stable states, thresholds, time lags, feedback loops or perhaps even irreversibility (Spears *et al.* 2012). Efforts to restore rivers to WFD good status have shown that a trajectory towards reference states of water bodies may be impossible to achieve due to the dynamic nature of river systems (Bouleau & Pont 2015). Again, this limits the effectiveness of using static ecosystem mapping to inform catchment management. ICM requires outputs which reflect the range of outcomes that management options may have on ecosystem service delivery.

2.3.5. Using Scenario Analysis

Scenario analysis is such a tool that has great potential for ICM as it can illustrate varying model states of a catchment. This may include accounting for possible temporal changes to policy and environmental drivers, and understanding how harder-to-predict, highly stochastic factors (e.g. world food market prices) can potentially affect future ecosystem service provision. There are a number of studies which combine trade-off analysis with scenario-based analyses such as determining the effects of changing policies, land cover or climate change (Bateman *et al.* 2013; Kirchner *et al.* 2015; Lawler *et al.* 2014; Nelson *et al.* 2008). Developing a number of land use scenarios using ecosystem service mapping tools may help inform policy-makers of potential trade-offs between different options for land use planning to improve catchment management (Zheng *et al.* 2016).

Scenario analysis may also help account for the large variations among stakeholder groups' views and preferences on catchment management and ecosystem service provisioning. Effective ICM requires input from local stakeholder groups to allow appropriate management that is based on local data, pressures and priorities, with further inputs from catchment scientists critical for ensuring that current understanding of catchment functioning is utilised.

2.3.6. Participatory Approaches

Participatory approaches to management of common resource pools are not a new concept (Ostrom 1990). However, in today's highly institutionalised top-down controlled social systems there is a need to relearn and reinvent bottom-up involvement to achieve more effective management of common resources and services, as conflicts may escalate when certain stakeholders are marginalised or ignored (Robbins 2004). Priorities of stakeholder groups vary depending on local values and pressures (Kaye-Zwiebel & King 2014), therefore policy-makers face the challenge of developing stakeholder-led, catchment-based approaches (McGonigle *et al.* 2012), which in turn can enable stakeholder groups to understand the pressures on their catchment system, assess their differing objectives and responsibilities, and help them consider possible synergies and conflicts of different management options. Involving stakeholders in decision-making throughout the planning process can make management options more inclusive, socially acceptable and maximise the likelihood of successful implementation of measures and strategies (Etienne et al. 2011; Knight *et al.* 2006; Oliver *et al.* 2017). Participatory approaches can also enable the inclusion of services such as aesthetics or cultural values within ecosystem service assessments, which may not be quantified using spatial mapping. Finally, such approaches can help build social capacities of catchments to make communities more responsive, resilient and capable (Kuhlicke *et al.* 2011).

Some participatory approaches use mapping to enable stakeholder-driven weighted ratings of spatial ecosystem service provision, in turn highlighting important areas of special social and ecological importance which conventional service mapping cannot detect (Bryan *et al.* 2011; Mahboubi *et al.* 2015). However, participatory approaches need to be underpinned by appropriate resources including sufficient time to explain methods, protocols and to build up stakeholder relationships, but when applied sensitively they can facilitate community-based catchment management by legitimising, analysing and representing local knowledge (McCall & Dunn 2012). More influential stakeholders and institutions may not commit to this concept of full cooperation, and party politics and lobbying pressures may further impede bottom-up decision-making. A further challenge for stakeholder participation is that it takes time and money to build meaningful relationships with stakeholders, which is often limited by a three year research funding cycle (Nancarrow 2005). Trust building of an effective stakeholder group can take years to develop, and trust can also erode over time (Menzel & Buchecker 2013). Although participatory approaches require significant time, resources and trust, they are a vital part of ICM and research can play a great role in developing engagement techniques which aid sustainable management of catchments.

2.4. Recommendations for Future Research

Addressing the significant global challenge of sustainable water and energy management while at the same time delivering food security for nine billion people by 2050 represents an exciting new frontier in catchment science (Foley 2011; Smith 2013). This will not be achieved by using a fragmented, piecemeal approach to catchment management. Instead, the lens of investigation needs to be focused to capture the entire catchment continuum and the complexity of environmental, ecological and social connectivity across the landscape. This requires the deployment of advanced environmental and social science tools grounded in fundamental and theoretical research to bridge traditional disciplinary boundaries, which can deliver strong applied and societal impact by maximising ecosystem service provision. This is driven, in part, by a policy and research landscape which increasingly recognises that water courses absorb pressures from their entire catchment area and must therefore be managed more holistically to increase benefits from catchment ecosystem services without compromising catchment functioning. Progress in this area has been evaluated in Sections 2–4, with significant developments in ICM and ecosystem service assessment noted. However, these concepts are still developing and have either not yet been put into practice, or matured sufficiently, to significantly improve sustainable management of catchments and their associated resources. A series of recommendations for future research are therefore proposed to help to promote effective ecosystem service provision via more integrated catchment management. The recommendations have been organised into a research agenda of short-term (0–5 years) and long-term (>5–10 years) opportunities.

2.4.1. Short-Term Research Priorities (0–5 Years)

Growing interest in WEF nexus science has led to a parallel increase in research to explore challenges and opportunities of managing trade-offs associated with decision-making within this critical nexus. Identifying the most pressing research questions needed to underpin WEF nexus science is therefore a clear priority, with co-designed research agendas beginning to emerge that combine viewpoints from researchers and business leaders in an effort to help companies manage their WEF nexus impacts (Green *et al.* 2015). Most research attention thus far has focussed on small-scale nexus interactions, such as N pollution from biofuel crops (Donner & Kucharik 2008), rather than exploring more efficient ways of sustaining water, energy and food security. The WEF nexus agenda also provides a strong rationale for energy and climate change policies to learn from the multiple and cumulative benefits that 'nexus-thinking' may deliver, particularly with respect to preserving freshwater ecosystems, such as wetland and floodplain restoration for carbon sequestration and water quality and biodiversity improvements, or energy generation from sewage (Pittock 2011).

2.4.1.1. Transdisciplinary Research and Stakeholder Engagement

The promotion of knowledge exchange mechanisms to foster more effective communication between scientists and decision-makers is becoming increasingly common in helping to tackle complex environmental challenges (Karpouzoglou *et al.* 2016). Catchment management has benefitted from recognition that such approaches help to build consensus and share wider perspectives to inform the decision-making process, but a more comprehensive understanding of the variability of viewpoints associated with decision-makers and beneficiaries of catchment management is needed, and across multiple spatial and temporal scales (Austin *et al.* 2016). Innovation in stakeholder engagement methodologies will help to underpin this need for improved understanding as participatory research continues to evolve to take advantage of more engaging and mutually beneficial approaches such as participatory modelling and the co-production of knowledge (Mahboubi *et al.* 2015; Whitman *et al.* 2015). More in-depth analysis of the characteristics of successful participatory modelling and research are clearly needed to help overcome the cultural, economic and technical constraints that can hinder effective engagement and participatory research. Participatory research should therefore focus on novel ways to exploit the benefits of social learning (Matthies *et al.* 2016). Wider deployment of structured engagement approaches should also be encouraged. For example, the Citizens' Jury technique is a useful approach to facilitate the combination of stakeholder and expert knowledge, which has already been successfully used in the context of catchment planning and risk management of microbial water pollution (Huitema *et al.* 2010; Fish *et al.* 2013). This technique has also been used to debate other contentious environmental issues, where it can provide a platform to facilitate social learning (Kenyon *et al.* 2003; Petts *et al.* 2001).

2.4.1.2. Ecosystem Service Assessment and Decision-Making Frameworks

To encourage the integration of stakeholders with catchment management decisions, existing ecosystem service assessment models and their outputs must be made more widely accessible. A particularly novel and interdisciplinary proposal is the 'gamification' of models and decision-support tools, made possible by the design of gaming interfaces as a user-friendly front-end to such tools, i.e., a graphic user interface. This would allow a large number of participants to 'play' and explore the underpinning science associated with the complexity of multiple impacts of decision-making which could, for example, allow a comparison of stakeholder accumulated trade-off preferences (Turner *et al.* 2016).

Exploiting ecosystem assessment studies to map wider arrays of ecosystem service provision in catchments is also a clear priority for the short-term future. These should aim to reflect the broad services that catchments provide, estimating multiple provisions in the WEF nexus as well as regulating services such as habitat and biodiversity provision, pollution and flood water buffering, and assess multiple improvements that management interventions may have on these. There is also a gap in researching the cultural services catchments provide to enhance people's well-being such as through landscape value, cultural heritage and recreational benefits. In doing so, a more comprehensive understanding of the likely interactions across ecosystem service provision should emerge, which in turn will help guide catchment management options that offer multiple benefits to wider society. Another important avenue of research is to understand how trade-offs and co-benefits spatially manifest themselves across terrestrial, freshwater and marine realms to assess potential synergies or conflict and inform sustainable management decisions (Adams *et al.* 2014).

Encouraging research to be more 'outcome driven' would offer advantages for the delivery of cost-effective environmental measures, which are likely to be implemented as part of integrated landscape management approaches, and capable of producing measurable benefits. Decision-making frameworks which incorporate 'outcome-oriented objective settings' can incorporate objectives such as maximising multiple ecosystem service provision or improving EU WFD ecological status as well as include socio-economic constraints, such as time, political context, governance and cost-effectiveness to select effective management options (Tulloch *et al.* 2015). Such frameworks are likely to improve over time with more innovative integration of primary data, ecosystem service mapping, and expert elicitation to select management options for their greatest return on investment or to predict the outcomes of different options on service provision (Joseph *et al.* 2009; Withey *et al.* 2012). This would certainly help to guide future research designed to consider the potential outcomes of a portfolio of different catchment management actions, whether it focuses on improving a single ecosystem service or aims to maximise multiple ecosystem service provision.

Of the large number of decision-making tools developed for environmental management, few are actually used to inform policy (Borowski & Hare 2007; van Delden *et al.* 2011). This can be due to lack of confidence in the product, lack of ease of use, and due to the length of time it can take to embed a tool within an organisation that may ultimately use it. To prevent this 'implementation gap' there should hence be active engagement between providers and users throughout the development of any tool to maximise its utility, acceptability and speed of uptake (Oliver *et al.* 2012). Involving stakeholders in the design of a decision support tool for visualizing *E. coli* risk on agricultural land, for instance, has helped to promote enthusiasm and understanding of the tool, and has enhanced its applicability (Oliver *et al.* 2017).

More in-depth reporting of model methodology from ecosystem service assessment studies is essential, as clear information on specific modelling parameters and indicators is often unreliable or missing (Seppelt *et al.* 2011). Furthermore, there have been calls for more defined methods and indicators for ecosystem service assessment to be made available to allow the reporting of consistent and comparable results (Lamarque *et al.* 2011). Promoting a more coordinated strategy would enable the potential reuse of research outputs by improving the comparability of studies, perhaps with more effective outcomes with respect to real-world improvements in catchments. This may be a significant challenge for the research community however, as most researchers are likely to favour their own approaches and methodologies. More generally, reporting practices and standards need to be introduced for measuring ecological, socio-cultural and economic values to increase comparability and transferability (de Groot *et al.* 2010).

Due to the lack of independent validation (inconsistencies among mapping approaches and little recognition of associated errors), land cover based proxy-maps can only ever be a crude estimation of spatial ecosystem service provision (Egoh *et al.* 2007; Schulp *et al.* 2014; Seppelt *et al.* 2011). Eigenbrod *et al*. (2010) first quantified the margin of error in land use based maps in the UK and showed that even good proxies could only show broad trends for ecosystem service provision, and were unsuitable to accurately show areas of multiple service provision or ecosystem service hotspots. Mapping must also communicate modelling uncertainty, as failing to incorporate this into mapping and decision-making may increase costs together with the likelihood of selecting unsuccessful management options (Tulloch *et al.* 2015). Acknowledging and estimating uncertainty in ecosystem service prediction maps should be highly encouraged at the research funding stage to increase the quality of ecosystem assessment studies.

Uncertainty estimation should also be integrated into other ICM tools. Certainty weightings can be incorporated into expert elicitation methodologies (Page *et al.* 2012), and setting upper and lower bounds on parameters can highlight best- and worst-case scenarios, rather than a single outcome. Bayesian Network models can present prediction uncertainties as probability distributions and such approaches have previously been used for ICM decision-support tools (Holzkämper *et al.*

2012). However, running uncertainty analyses for all subcomponents of an integrated model is extremely time-consuming (Lerner *et al.* 2011). Bayesian Networks can also be used to develop scenario-based analysis methods, which incorporate the likelihood of scenarios occurring, and reveal effects of continuous scenarios as opposed to a handful of discrete options (Dong *et al.* 2012). Computational power is growing at a rapid rate, increasing the feasibility of uncertainty analysis in the future, which will allow models to explore spatial and temporal synergies and trade-offs and assess how ecosystem service provision vary in space and time. Models will, however, still be limited due to the uncertainties around our understanding of ecosystem services and how they interact.

2.4.2. Long-Term Research Priorities (>5–10 Years)

Institutional and structural barriers, such as a lack of commitment to ICM by decision-makers, limited time and resources, ineffective communication at the science-policy interface and poor internal collaboration need to be tackled to allow successful ICM. Research should aim for greater integration across political and scientific scales, such as integrating catchment and marine objectives (Álvarez-Romero *et al.* 2015), but also across national borders to facilitate the best possible outcomes (Levin *et al.* 2013). There is also scope for more studies to integrate local stakeholders into the research process, from design to implementation, by co-producing ICM strategies. It will be a significant challenge to integrate land management policy at national scales, despite a recognition of the need for policies that deliver on a range of ecosystem services (McGonigle *et al.* 2012). Indeed, in most countries, many of the management decisions around the WEF nexus and across terrestrial, freshwater and marine realms are highly fragmented and do not consider potential trade-offs or multiple benefits between services and realms.

The existing evidence-base of published ecosystem assessment studies has a particular focus on the more developed nations of the world, with particular paucity of published research coming from Africa (de Araujo Barbosa *et al.* 2015). ICM linked to an ecosystem services framework would represent a significant research opportunity in less developed nations, especially as this would reduce their reliance on expensive engineering structures (Green *et al.* 2015). There is, however, a wealth of studies around PES schemes in developing countries (especially from Latin America), and there are exciting opportunities for the two-way exchange of results and research approaches from developing and developed countries (Schomers & Matzdorf 2013). Both Costa Rica and Mexico are, for example, developing additional financing sources from ecosystem service beneficiaries and are aiming for more targeted and differentiated payment schemes (Wunder *et al.* 2008). Adapting such approaches for EU agri-environment schemes may make these schemes more effective at improving ecosystem service provision, and be more cost-efficient. However, it is vitally important to incorporate the cultural and institutional differences into such method development, particularly when transferring PES approaches from countries with strongly developed legal frameworks to countries with a weak legal and institutional environment (Schomers & Matzdorf 2013). ES modelling and stakeholder engagement tools usually assume a stable government, infrastructure, available data and a willing and educated stakeholder pool. To successfully develop ICM in places with the greatest need will require a completely new approach, with the development of novel and innovative tools that work with local environmental, socio-economic and cultural constraints.

Finally, the longitudinal analysis of stakeholder perceptions, and how views might change, offers a particularly novel angle to the social science dimensions of catchment management. For example, the medical and health sciences often use cohort studies to track how different lifestyle choices can impact on health and well-being of the public, and in some cases such cohort studies can track results for extended periods, possibly decades. Transferring this concept to track temporal shifts in stakeholder and/or catchment citizen perceptions over time would be an interesting prospect. Participants of such a cohort study would presumably be exposed to and experience different 'catchment lives', which may impact on their perceptions and values of ecosystem service provision. The funding of such longitudinal study would clearly be challenging, especially given that longer timescales of tens of years would be of particular interest to monitor how shifts in cohort perceptions vary during their exposure to wider catchment understanding. Nonetheless, such a study would certainly represent frontier interdisciplinary research to better our understanding of the socio-ecological complexity of catchment systems.

2.5. Conclusion

Integrating complex stakeholder relationships and ecosystem service interactions into the process of ICM represents a significant challenge but also an exciting opportunity. The ecosystem service concept is useful to raise awareness among catchment managers and policy-makers of the need to maximise multiple ecosystem service provision, as opposed to selecting management options primarily to increase production of a few selected tangible services. Multidisciplinary and multi-stakeholder ecosystem service assessment can reveal hidden costs of managing an ecosystem simply for provisioning services, and help select management options for multiple ecosystem service provision to secure water, food and energy security while protecting the environment and human health and well-being. The practice of sustainable catchment management is now at a pivotal juncture where it faces the challenge of meeting an increasing amount of often competing demands on the services provided by catchment systems, but is also presented with a number of innovative and emerging research tools for deployment to begin to address that challenge. With careful assessment and continued efforts to deliver quality, cutting-edge research, the opportunities will outweigh the challenge.

3. Quantifying stakeholder understanding of an ecosystem service trade-off

Published in Science of the Total Environment 2019

Manuscript details:

Stosch, K. C., Quilliam, R. S., Bunnefeld, N., & Oliver, D. M. (2019). Quantifying stakeholder understanding of an ecosystem service trade-off. *Science of the Total Environment*, *651*, 2524-2534.

DOI: https://doi.org/10.1016/j.scitotenv.2018.10.090

Author contributions:

K. C. Stosch conceived and planned the experimental work with advice and supervision from D. M. Oliver, R. S. Quilliam and N. Bunnefeld. K. C. Stosch carried out all experimental work and data analysis, and wrote the manuscript. D. M. Oliver, R. S. Quilliam and N. Bunnefeld suggested improvements to the manuscript. Anonymous reviewers suggested improvements to the final manuscript.

3.1. Introduction

Sustainable management of natural resources is challenged by social and environmental drivers such as rapid population growth and changing climatic regimes. In turn, ecosystem service provision is under pressure in many regions where there are competing demands on environmental resources, leading to interactions and trade-offs within socio-ecological systems (Cumming *et al.* 2014). Thus, ecosystem services are spatially heterogeneous and temporally dynamic, responding to human and environmental pressures but also shifts in other ecosystem services. The ecosystem service concept has therefore gained recognition as an approach for addressing interactions within socio-ecological systems, both by research and policy-practitioner communities and those with a responsibility for land-based decision-making (Costanza *et al.* 2017; Ma *et al.* 2016).

Interdependency between ecosystem services presents a principal challenge for sustainable landscape management (Cordingley *et al.* 2016). Interactions between provisioning and other ecosystem services are generally dominated by negative correlations or trade-offs, e.g. a decrease in runoff water quality with increased livestock grazing densities (Austrheim *et al.* 2016), while synergies are often found between regulating and cultural services (Lee & Lautenbach 2016; Lin *et al.* 2018), such as the increase in biodiversity, pollination and biological pest control from flower strip planting (Westphal *et al.* 2015). Changes in land management to enhance a single service may often cause calculated but also inadvertent trade-offs, especially at larger spatial and temporal scales beyond those of the immediate management concern (Rodríguez *et al.* 2006). Agricultural intensification can, for example, negatively impact on pollinator diversity, which in turn can affect the yield of pollinator-dependent crops (Deguines *et al.* 2014). Trade-offs in river catchments are often expressed downstream of management decisions, and can lead to conflict between upstream and downstream users (Asquith et al. 2008). Downstream trade-offs maybe so severe that they become irreversible (Bennett et al. 2009), such as degraded aquatic ecosystems, which can, despite extensive restoration efforts, fail to recover to their original reference state (Bernhardt & Palmer 2011). Therefore, investments in conservation, restoration and sustainable natural resource use are increasingly seen as 'win-win' opportunities, generating substantial ecological, social and economic benefits(de Groot *et al.* 2010).

Multiple services, or bundles of ecosystem services, are often mapped to establish whether trade-offs exist based on co-occurrence (Raudsepp-Hearne et al. 2010; Turner et al. 2014). This has led to an increased interest in the understanding and optimisation of ecosystem services for environmental management, with the aim of improving the delivery of regulating and cultural services without compromising provisioning services (Austin *et al.* 2016; O'Sullivan *et al.* 2017; Weijerman *et al.* 2018). Catchments are, however, socio-ecological systems, and therefore a trade-off does not only arise due to relationships between ecosystem services, but also due to diverging stakeholder perceptions on ecosystem service provisioning (Martin-Lopez *et al.* 2012). Different stakeholder typologies may express varying preferences for ecosystem services, depending on their knowledge, values and connections to the landscape (Lamarque *et al.* 2011; García-Nieto *et al.* 2015).

Stakeholders involved in agriculture in water-limited areas, for instance, are more aware of the ecosystem service benefits of maintaining water flows (Castro *et al.* 2014). Social contexts such as livelihoods, interests and traditions influence stakeholder perception of ecosystem services, which may lead to conflict among opposing stakeholder groups, i.e. between farmers and conservationists (Cebrián-Piqueras *et al.* 2017).

Combining trade-off analysis with stakeholder engagement offers potential to facilitate effective knowledge exchange between decision-makers, while also capitalising on important expertise and understanding that would be otherwise missed from trade-off analysis alone (Galafassi *et al.* 2017), as well as highlighting stakeholder typology differences in ecosystem service perception (Darvill & Lindo 2016). Including surveys as part of ecosystem service analysis, for instance, can help to capture the complexity of socio-ecological systems by incorporating stakeholder values and identifying drivers of change (Andersson *et al.* 2015; Garcia-Llorente *et al.* 2015). Participatory mapping techniques can aid understanding of the spatial distribution of social benefits, especially for cultural services, which are difficult to estimate (Canedoli *et al.* 2017; Reilly *et al.* 2018). The use of participatory approaches is therefore vital for including the social demand of ecosystem service trade-offs, which is often neglected, and hence may avoid potential conflict of natural resource use and management (García-Nieto *et al.* 2013).

Another technique that integrates the supply and demand side of ecosystem service trade-offs is the production possibility frontier (PPF) concept. The PPF delineates the biophysical relationship between two ecosystem services and represents the maximum values they may attain within that trade-off (Cavender-Bares *et al.* 2015; see section 3.2.1 for a more detailed description). The utility function indicates the point along the PPF where the utility of the two ecosystem services is maximised for a stakeholder. It is difficult to estimate PPFs and particularly utility functions of an ecosystem (Lester *et al.* 2013), but there are studies that approximate the PPFs of services between two (Lang & Song 2018) or multiple ecosystem services (Lautenbach *et al.* 2013). There is, however, considerable scope for including utility functions in trade-off analysis to characterise the social demand of ecosystem service interactions (Cord *et al.* 2017). The use of participatory research to assess perceptions of the PPF of a trade-off and associated utility functions can reveal differences in stakeholder priorities concerning more complex ecosystem service interactions.

To our knowledge, there are no previous studies that assess stakeholder views on the shape of a PPF, or their perceptions on stakeholder utility functions within a trade-off. In response, we developed a novel stakeholder engagement methodology which elicits the perception of four key stakeholder groups working in land and water management. We quantified their assessment of both the shape and the uncertainty around the PPF in a trade-off between agricultural intensity and freshwater ecological health. We further quantified how participants perceived the utility functions of different stakeholder groups within that trade-off. Our objectives were to investigate stakeholder views to: (1) define the nature of, and the uncertainty associated with, a specific water and land management trade-off; (2) estimate stakeholder prioritisation of the trade-off; (3) quantify how views varied in different catchments and across different stakeholder groups; and (4) assess the practical relevance of this participatory methodology for land and water management planning and decision-making.

3.2. Materials and methods

3.2.1. The 'production possibility frontier' (PPF) concept

Depending on the biogeophysical constraints on a pair of ecosystem services, together with how they are managed, the PPF may take a number of different forms which are often non-linear in nature (Fig. 3.1; Koch *et al.* 2009). In an exponential decline PPF, the ecosystem service on the x-axis correlates with a sharp decrease even at small increases of the other ecosystem service (Fig. 3.1c). In contrast, the response is initially more resilient on the threshold (Fig. 3.1e) and logistic decay (Fig. 3.1f) function with a rapid decline once a threshold is passed. With the intermediate disturbance function PPF, moderate increases in one ecosystem service have a synergistic effect on the other, but larger increases are detrimental to it (Fig. 3.1d).

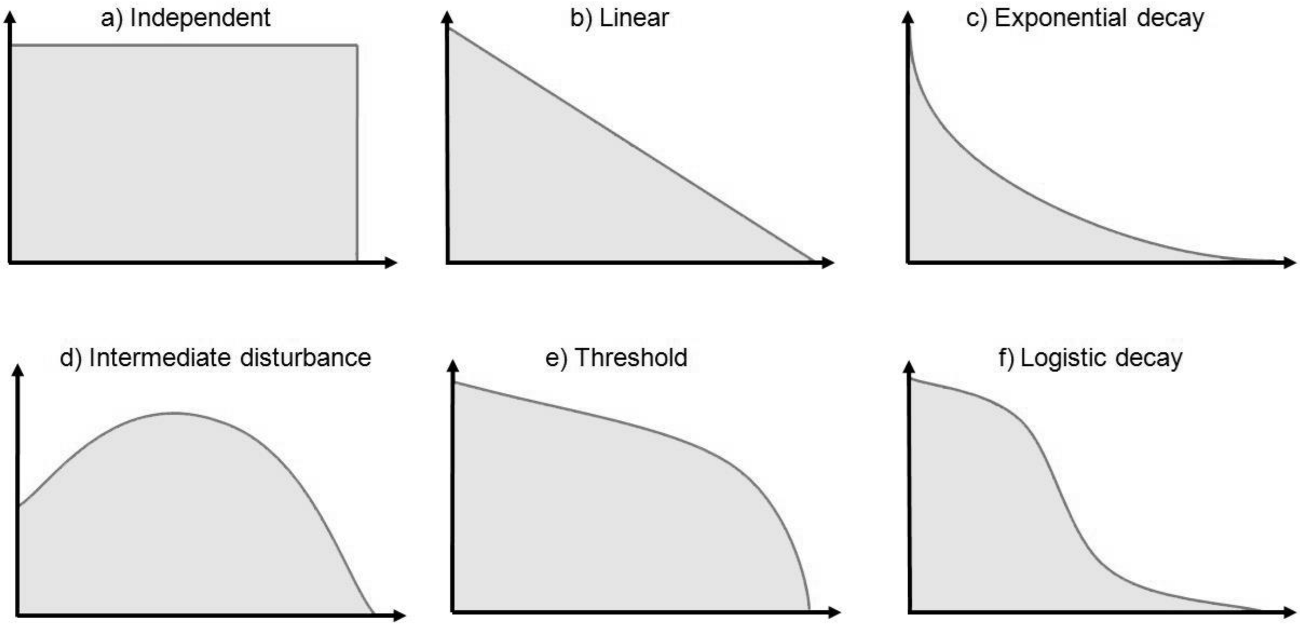


Fig. 3.1: Illustrating the possible forms the trade-off between two ecosystem services may take: (a) independent, (b) linear, (c) exponential decay, (d) intermediate disturbance function, (e) threshold relationship, and (f) logistic decay (Koch *et al.* 2009).

Isoclines of stakeholder utility values are plotted over the PPF function (Fig. 3.2a and b), which represent the utility value that a stakeholder places on the ecosystem services in a specific trade-off. The utility function of a given stakeholder is the point where the isoclines meet the PPF, and represents where the trade-off should be balanced to maximise utility for the stakeholder. When plotting multiple trade-off preferences, the distance between the utility functions can highlight potential conflict between stakeholders' positions on how a trade-off should be managed to balance the preferences of multiple stakeholders. Taking the example of the trade-off between agricultural yield and downstream water quality: although the PPF represents the maximum output within a trade-off scenario (Fig. 3.2a), the area under the PPF curve may be increased by implementing management that does not negatively impact on yield while preserving water quality, such as through efficient fertiliser use (Fig. 3.2c; Ewing & Runck 2015). In turn, this then allows the utility values of both stakeholders with competing demands to be improved.

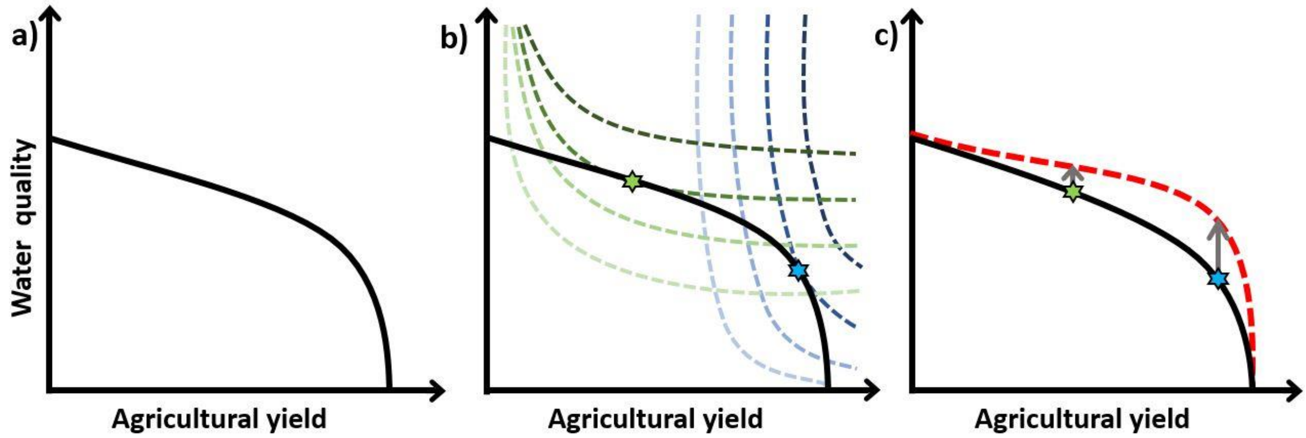


Fig. 3.2: (a) The 'production possibility frontier' (PPF; black line) of a trade-off between two ecosystem services delimits its biophysical constraints. (b) Stakeholder preferences within the trade-off, called 'utility functions' (green and blue star) are constrained by the PPF and by the utility value of the stakeholders indicated by the isoclines (green and blue dotted lines). (c) The PPF may be altered by changing the management of the ecosystem, which may benefit both stakeholders. Adapted from King *et al.* (2015).

3.2.2. Study catchments and stakeholder sample

Three catchments from across Scotland were selected on account of their diverse geomorphologies, land cover types, stakeholder communities and land and water management pressures. The River Spey in the north-east, the South Esk in the east and the River Ayr catchment in the south-west of Scotland (Fig. 3.3). The catchments vary in size from ~ 600 km^2 (South Esk and Ayr) to just under 3000 km^2 (Spey). Moors and heathland is the most dominant land cover type in the Spey (29%; Table 3.1) and the Esk catchment (33%), followed by sparsely vegetated land in the mountainous areas of the Spey (23%) and arable land in the Esk catchment (31%). Dairy production is a key local industry in the Ayr catchment with pasture accounting for 39% of the land cover.

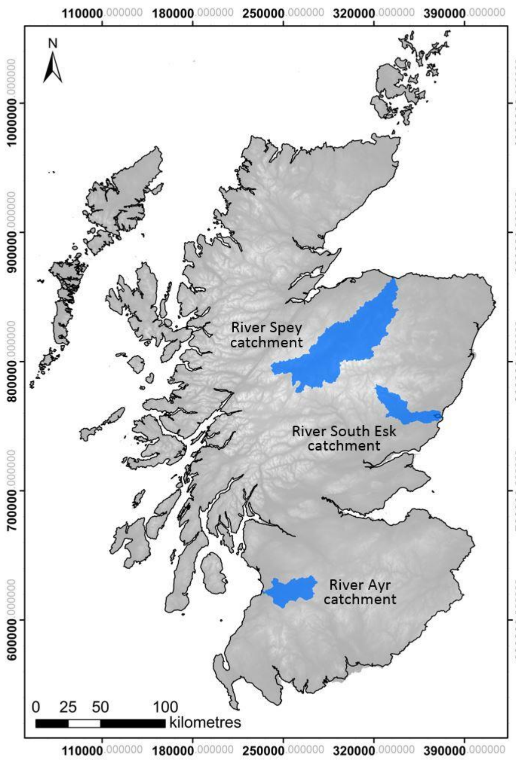


Fig. 3.3: The three study catchment areas: The River Spey in the north-east, the South Esk in the east and the River Ayr catchment in the south-west of Scotland.

In general, the uplands of the three catchments are dominated by rough grazing, commercial forestry, and sporting estates, while the lowlands accommodate arable land and improved grazing. Tourism and angling represent important local industries, with whisky production also being significant, particularly in the Spey. There are competing pressures on water resources in all three catchments via diffuse pollution from farming practices and point source inputs from sewage discharge, in addition to abstraction for potable water, large hydropower schemes, food and drink manufacture and irrigation.

Table 3.1: Land cover types in the three study catchments as a percentage of overall area covered (rounded to the nearest whole number).

Land cover type	Spey catchment	Esk catchment	Ayr catchment
Moors & heathland	29%	33%	11%
Coniferous forest	16%	8%	9%
Pastures	9%	12%	39%
Sparsely vegetated areas	23%	0%	0%
Natural grasslands	9%	10%	14%

Arable land	2%	31%	7%
Peat bogs	7%	1%	10%
Transitional woodland-shrub	3%	1%	2%
Broad-leaved forest	2%	1%	1%
Urban areas	1%	1%	2%

A total of 43 stakeholders participated in the study, completing an engagement exercise on PPF characterisation for a specific trade-off within their respective catchments. Three to five individuals from four key stakeholder groups were interviewed in each of the three study catchments. The four stakeholder groups were selected through a preliminary desk-based exercise that ranked the importance of the stakeholder groups for land and water management, and their influence on management decisions. Participants belonged to one of four key stakeholder groups: Environmental Regulators (*n*=12; all staff from the Scottish Environment Protection Agency), Water Industry Staff (*n*=9; all from Scottish Water, Scotland's public water and wastewater company), Catchment Scientists (*n*=11; from universities and research institutes across Scotland) and Farm Advisors (*n*=11; from the National Farmers Union Scotland, as well as independent farm consultants). Criteria for selection of participants was: (i) evidence of experience in their respective catchment, e.g. an individual was required to have worked for at least a year in the catchment, or written a publication or report linked to the catchment; and (ii) expertise on land and water management issues. Participants were initially identified through a desktop search with additional stakeholders identified via recommendations from initial stakeholders.

We investigated the trade-off between agricultural intensity and a measure of aquatic health, because diffuse pollution from agriculture continues to challenge the ecological status of many waterbodies in Scotland and the UK, as regulated under the EU Water Framework Directive (WFD). Ecological status, as defined by the WFD, is a robust measure of aquatic ecosystem health, integrating a number of physical, chemical and biological indicators. Ecological status was therefore used as a measure in our study because it is a well understood term amongst the four stakeholder groups, and has direct policy implications. Implicit within this measure are the delivery of a number of ecosystem services, as improved ecological status will lead to increased provisioning services, such as water supply and fish stocks, as well cultural services, such as tourism and recreation. Agricultural intensity was selected, in preference to the ecosystem service of a particular agricultural yield, as this measure includes other land management practices such as livestock farming, slurry spreading and silage production and is therefore much more applicable to a variety of river catchments.

3.2.3. Method design and data collection

The interviews were conducted one-to-one using a tablet computer as part of a mixed method survey, integrating qualitative and quantitative data and approaches from environmental science and social science research. Participants were presented with a Windows PowerPoint document on the tablet computer, showing a blank trade-off graph with agricultural intensity on the x-axis (ranging from 0 to 1) and ecological status on the y-axis (on a scale between 0 and 1). The WFD measure of ecological status ranges from high ecological status, to good, moderate, poor and bad as the ecological quality of a waterbody deteriorates.

The interviewer explained the axes to the participant and asked what they perceived the shape of the trade-off between those two factors to look like in their river catchment, under the current land management practices in their respective catchment and disregarding other management that may impact on ecological status, such as urban developments. The interviewer would then present the participants with the next PowerPoint slide depicting four possible shapes (Fig. 1b, c, e or f), and were asked to select the one that they considered best represented the true PPF in their catchment. The independent and intermediate disturbance shapes were not given as an option, as there is evidence that increased agricultural intensity negatively impacts the ecological status of aquatic ecosystems (Stoate *et al.* 2009). The interviewer would then skip to a PowerPoint slide that had that shape selected on the trade-off graph. After identifying a shape to associate with the trade-off, participants were then asked to select one of three pre-drawn 95% confidence intervals around the PPF, which could either be of small, intermediate or large uncertainty. The interviewer would flick between the three options for their chosen shape to aid their decision-making. This provided a measure of how confident they were that their chosen PPF corresponded to the true underlying PPF in their catchment, and of the underlying variability in the catchment system.

After choosing the PPF and the confidence intervals, participants were asked to consider how they perceive utility functions to vary across different stakeholder typologies. Here participants were presented with coloured circles on the tablet (which corresponded to each of the four stakeholder groups), to place on the PPF at the point where they perceived maximum utility for each group. The size of the utility functions could be enlarged by the participants, allowing a range of maximum utility to be selected for each stakeholder group instead of selecting one point along the PPF. The interviewer explained that enlarging utility functions could hence include an estimate of the uncertainty in identifying the true mean of the stakeholder group's utility function, but also to account for within stakeholder group variation of utility functions. Finally, participants were given the opportunity to review the figure and ensure their response accurately represented their views.

After completing the first exercise, stakeholders were asked to complete the exercise a second time, however this time the shape of the trade-off was pre-determined and all participants were asked to place utility functions for the four stakeholder groups on the same PPF (Fig. 1e). The threshold PPF was selected here, due to findings from Ewing and Runck (2015) that this shape represented the relationship between agricultural yield and a measure of water quality (nitrate concentrations), in their study on corn production in the mid-western United States. Therefore, each participant completed two figures as outputs, (a) one PPF of their choice including confidence intervals and four utility functions and (b) one threshold PPF with four utility functions. This allowed better comparison of utility functions between participants as responses would be more comparable when recorded on the same PPF. Furthermore, responses from participants that selected the threshold PPF in the first exercise could then be used as a control response to assess the accuracy of the placement of the utility functions when repeated.

3.2.4. Analysis

The responses from all participants were converted to numerical values by measuring the distance to the start of the utility functions on the x-axis and the diameter of their utility function to the nearest millimetre after ensuring the plots were standardised in terms of their scale on the tablet computer. Both the measurements of utility function starting position and diameter were scaled to values from 0 to 1 by dividing values by the total length of the x-axis after which basic descriptive statistics were obtained and statistical analysis undertaken using SPSS version 23 (IBM 2012). To compare responses among the different catchments and stakeholder groups a non-parametric statistical test (Kruskall Wallis) was used, as variances were often significantly different per Levene's homogeneity of variances test. As 16 participants chose the threshold PPF in the first exercise, which was also the PPF that all stakeholders responded to in the second exercise, their responses for the utility functions could be used as a control. For those responses, pair-wise comparisons were made between the utility functions from the first and second exercise using a Wilcoxon Signed Rank Test. The same test was used to compare within and between stakeholder group responses. Pearson's Chi-Squared Test of Association was used to analyse the association between the PPF and confidence intervals that were selected and which stakeholder typology the respondents belonged to. The 'exponential decay' and 'linear' functions were chosen infrequently by participants and those typologies were therefore categorised as 'others' for the purposes of statistical comparison of their count data with the 'logistic decay' and 'threshold curve' responses. Similarly, only the results for 'intermediate' and 'large' uncertainty intervals were compared, as counts for 'small' confidence intervals were insufficient for statistical analysis. Rstudio software version was used to produce the bar plot charts (RStudio 2016).

3.3. Results

3.3.1. Selection of the PPF and confidence intervals

Most stakeholders selected either the logistic decay (40%) or the threshold function (37%) to describe the shape of the PPF in their catchment. Four participants from the Farm Advisor stakeholder group, however, did not agree with any of the four shapes, as two of them thought the PPF would follow more of an intermediate disturbance curve. Two other Farm Advisors agreed it was a threshold relationship, but that it would never reach bad ecological status even at the highest agricultural intensities. There was no significant association between the PPF function selected and the stakeholder group or the catchment that the participant was associated with (see Table 3.2 for a summary of all the statistical outputs). However, most Environmental Regulators (67%) selected the logistic decay, while most Farm Advisors (88%) selected either the threshold curve or did not agree with any of the shapes offered. The confidence intervals chosen by stakeholders were mostly the intermediate (49%) or large (44%) confidence intervals and there was no significant association between the uncertainty selected and the stakeholder group the participant belonged to. However, Catchment Scientists predominantly chose large confidence intervals (73%) while Environmental Regulators were more likely to select intermediate uncertainty around the PPF (69%). The other two stakeholder groups selected both intermediate and large confidence intervals at equal proportions with 45% of Farm Advisors and 44% of Water Industry Staff choosing intermediate uncertainty and 45% of Farm Advisors and 44% of Water Industry Staff selecting large uncertainty.

Although the surveys were carried out across three diverse river catchments, no statistically significant differences were found between the catchments in any of the measures. Hence, data were aggregated and only differences among stakeholder typologies are presented.

Table 3.2: Summary of all the statistical testing undertaken in the study.

Variables compared	Statistical test	Test statistic	Value	DF	P-value
PPF shapes and confidence intervals selected by stakeholder group and catchment					
PPF selected & Stakeholder typology	**Chi-squared Test of**	Pearson	9.162	6	>0.05
PPF selected & Catchment	**association**	Pearson	3.237	4	>0.05
Uncertainty selected & Stakeholder typology		Pearson	6.644	3	>0.05
Uncertainty selected & Catchment		Pearson	0.957	2	>0.05
First and control response of utility function placement for each stakeholder group (Fig. 3a)					
Environmental Regulators	**Wilcoxon**	Wilcoxon statistic	45.0	15	>0.05
Catchment Scientists	**Signed Rank**	Wilcoxon statistic	42.5	15	>0.05
Farm Advisors	**Test**	Wilcoxon statistic	93.0	15	>0.05
Water Industry Staff		Wilcoxon statistic	62.0	15	>0.05
First and control response of utility function diameter for each stakeholder group (Fig. 3b)					
Environmental Regulators	**Wilcoxon**	Wilcoxon statistic	99.5	14	**<0.05**
Catchment Scientists	**Signed Rank**	Wilcoxon statistic	84.0	13	**<0.01**
Farm Advisors	**Test**	Wilcoxon statistic	66.0	12	**<0.05**
Water Industry Staff		Wilcoxon statistic	84.5	14	**<0.05**
Position of utility function of own group compared to response of other groups (Fig. 6a &b)					
On PPF chosen by stakeholder					
Environmental Regulators	**Wilcoxon**	Wilcoxon statistic	12.0	10	>0.05
Catchment Scientists	**Signed Rank**	Wilcoxon statistic	41.5	9	>0.05
Farm Advisors	**Test**	Wilcoxon statistic	25.0	9	>0.05
Water Industry Staff		Wilcoxon statistic	33.0	6	**<0.05**
On threshold PPF					
Environmental Regulators		Wilcoxon statistic	45.0	10	**<0.01**
Catchment Scientists		Wilcoxon statistic	21.0	9	>0.05
Farm Advisors		Wilcoxon statistic	62.0	9	**<0.01**
Water Industry Staff		Wilcoxon statistic	36.0	6	**<0.05**
Difference in utility function placement between typologies: Kruskall-Wallis Test (Fig. 7)					
On PPF chosen by stakeholder	H-value	Adjusted for ties	175.96	9	**<0.001**
Utility function positioning for the four stakeholder typologies: Kruskall-Wallis Test (Fig. 4)					
On PPF chosen by stakeholder	H-value	Adjusted for ties	59.83	3	**<0.001**
On threshold PPF	H-value	Adjusted for ties	36.50	3	**<0.001**
Utility function positioning by respondent's stakeholder group: Kruskall-Wallis Test (Fig.5)					
On PPF chosen by stakeholder					
Environmental Regulators	H-value	Adjusted for ties	2.08	3	>0.05
Catchment Scientists	H-value	Adjusted for ties	1.20	3	>0.05
Farm Advisors	H-value	Adjusted for ties	1.87	3	>0.05
Water Industry Staff	H-value	Adjusted for ties	6.24	3	>0.05
On threshold PPF					
Environmental Regulators	H-value	Adjusted for ties	15.91	3	**<0.001**
Catchment Scientists	H-value	Adjusted for ties	5.87	3	>0.05
Farm Advisors	H-value	Adjusted for ties	13.98	3	**<0.01**
Water Industry Staff	H-value	Adjusted for ties	16.98	3	**<0.001**

3.3.2. *Utility function responses*

When comparing the two responses of those participants who selected the threshold PPF in the first exercise (n=16), there was no significant difference in the position that the participants placed the utility functions on the threshold curve for the repeated PPF exercise (Fig. 3.4a), although their diameter was significantly smaller (Fig. 3.4b).

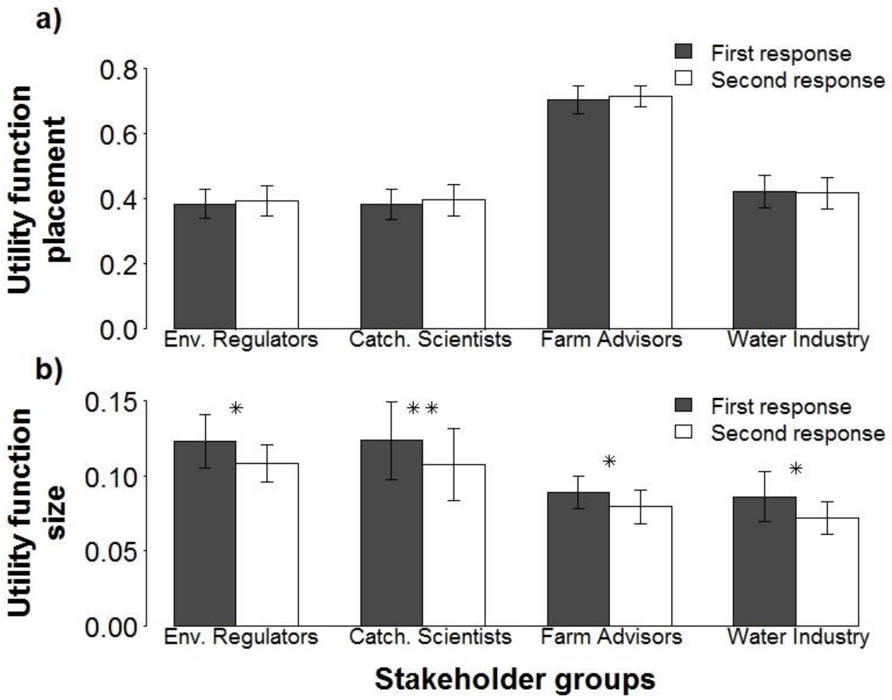


Fig. 3.4: Differences between (a) the position, and (b) the size of the utility functions from those participants (n=16) that used the threshold function both for their first (black) and second (white) response. Significantly different pairs are given at p<0.05* and p<0.01**. Error bars indicate ± 1 standard error.

When collating all responses from stakeholders, the combined PPF from the first exercise (Fig. 3.5a) represented an intermediate shape between the two dominant responses (logistic decay and threshold curve) and its confidence intervals fell between intermediate and large, as those were the two most prevalent replies.

In both the first (Fig. 3.5a) and the second exercise (Fig. 3.5b), the utility functions of the four stakeholder groups were identified as being significantly different from one another (p<0.001, H=59.83 and 36.50 respectively). In exercise 1 (Fig. 3.5a) the utility functions for Water Industry

Staff, Environmental Regulators and Catchment Scientists (in that order) were all located in close proximity to one another at around 0.85 for ecological status and 0.45 for agricultural intensity, while utility functions for the farm advisory group were positioned towards greater agricultural intensity (~ 0.6).

Utility functions on the pre-defined threshold PPF in the second exercise (Fig. 3.5b) delivered consistent rank ordering of the four stakeholder groups with the first exercise. The utility functions were, however, shifted towards greater agricultural intensity while remaining at a similar ecological status, with the Farm Advisors now located at an agricultural intensity ~0.75 to 0.8. In both exercises the utility function for the Farm Advisors were placed on the area of the PPF curve where its slope started decreasing, but before the rapid decline of ecological status.

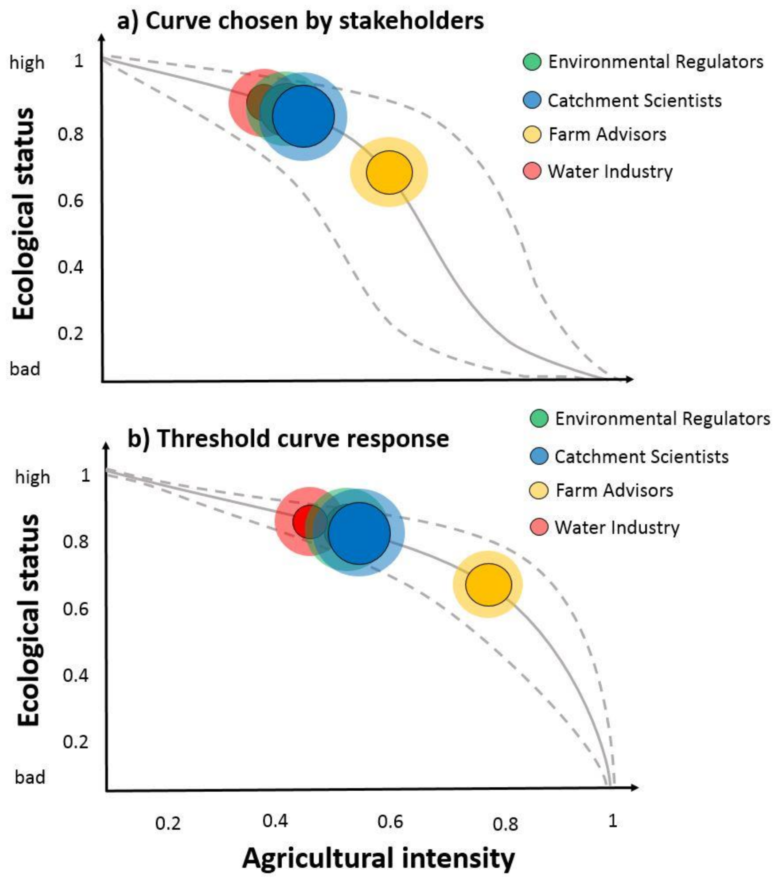


Fig. 3.5: Mean stakeholder responses of the four stakeholder groups' utility functions. The solid circles indicate where the four stakeholder groups were perceived to prioritise the trade-off (halos indicate + the standard error). The participants responded on a PPF curve (a) chosen by themselves, and (b) on the threshold PPF curve.

3.3.3. *Comparing responses depending on stakeholder typology*

When stakeholders had to consider how they expected other stakeholder groups would perceive PPF functions, utility functions were placed differently depending on which stakeholder group the participant belonged to. This was the case on the threshold PPF in the second exercise (Fig. 3.6), however not when comparing responses from the first exercise where PPFs differed. Neither did utility functions differ significantly between the three study catchments in either exercise 1 or 2. In the second exercise, responses by Catchment Scientists were most similar to the mean (Fig. 3.6b), while Water Industry Staff placed their own utility function at higher ecological status (Fig. 3.6d). Compared to the mean, Environmental Regulators estimated the utility functions to be at higher agricultural intensity (Fig. 3.6a) while the Farm Advisors reported utility functions towards lower agricultural intensity (Fig. 3.6c).

Only the utility functions of Catchment Scientists were not perceived differently by the four stakeholder typologies. The utility functions of Farming Advisors were placed at significantly higher agricultural intensities by Environmental Regulators and significantly lower by Farm Advisors ($p<0.05$, $H=13.98$). Utility functions for Environmental Regulators and Water Industry Staff were also perceived differently depending on the group affiliation of the respondents ($p<0.001$, $H=15.91$ and 16.98 respectively).

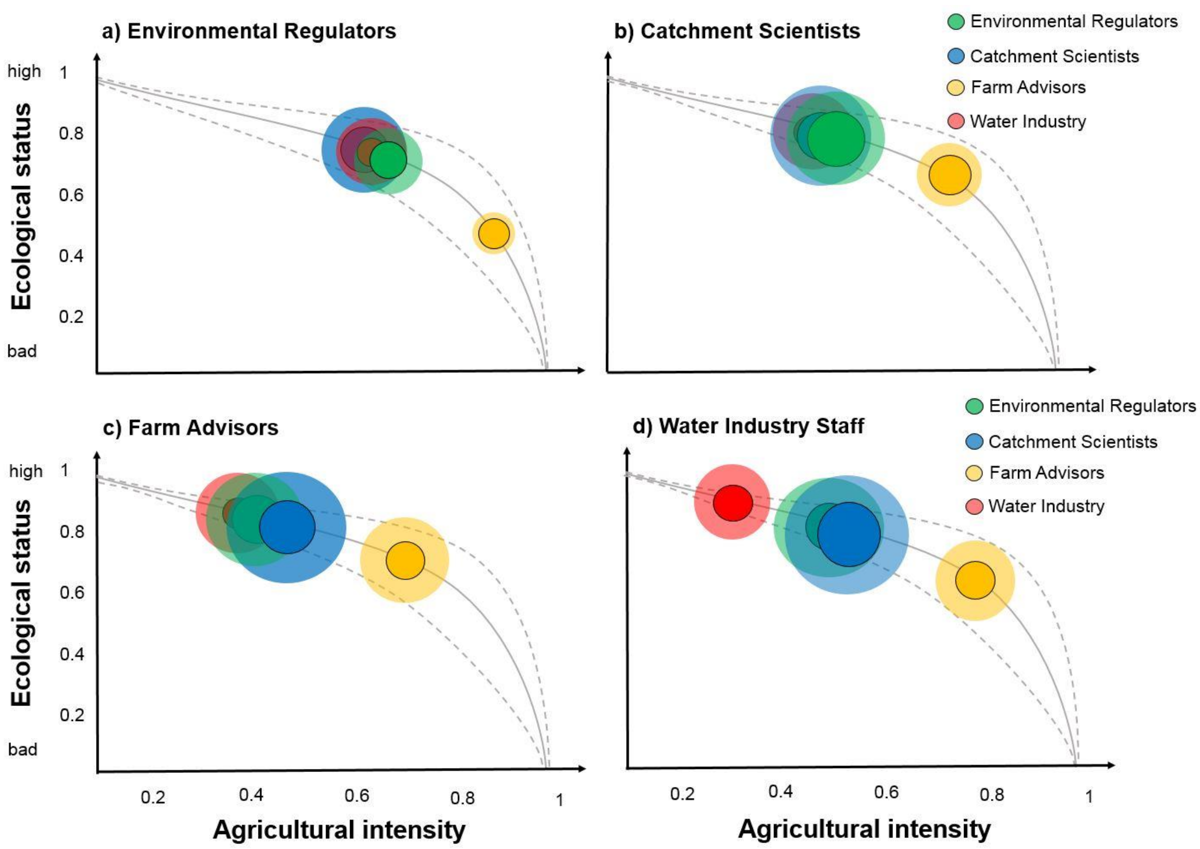


Fig. 3.6: Mean responses on the threshold PPF curve, by each stakeholder group: (a) Environmental Regulators, (b) Catchment Scientists, (c) Farm Advisors, and (d) Water Industry Staff. The solid circles indicate the perceived trade-off prioritisation of the four stakeholder groups (halos indicate + standard errors).

When comparing how participants viewed the utility functions of their own stakeholder group, as opposed to how the other three groups estimated them, a number of significant differences were identified (Fig. 3.7). Water Industry Staff scored their own utility functions at significantly higher ecological status compared to other groups' perceptions, both when they chose their own PPF ($p<0.05$, $W=33.0$), and particularly, on the threshold PPF ($p<0.05$, $W=36.0$). On the threshold PPF, Farm Advisors also scored their own utility functions at significantly lower agricultural intensity compared to others ($p<0.01$, $W=62.0$), while Environmental Regulators placed their own utility functions at significantly higher agricultural intensity compared to others ($p<0.05$, $W=45.0$). When comparing the mean differences of all utility function placements between stakeholder groups, the largest difference was between Environmental Regulators and Farm Advisors, while the responses of Catchment Scientists were most similar within their own group (Fig. 3.8; $p<0.001$, $H=175.96$). Utility function placement by Environmental Regulators was also more similar within their group while

Farm Advisors and Water Industry Staff differences within their own group were more similar to the mean difference in utility function scoring.

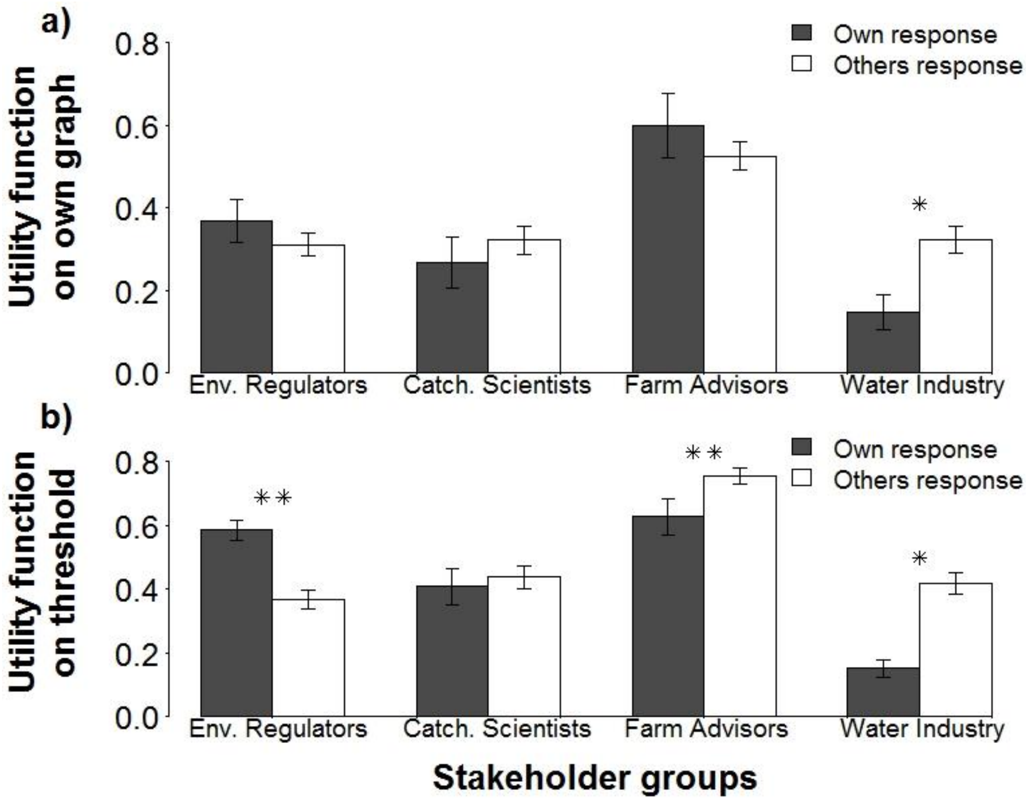


Fig. 3.7: Differences between the position of the utility functions on the x-axis of the trade-off graph, depending on whether they estimated their own group (black) vs. when others identified their stakeholder group (white), on both their first response using the graph chosen (a) by themselves, and (b) on the threshold curve. Significantly different pairs are given at p<0.05* and p<0.01**. Error bars indicate ± 1 standard error.

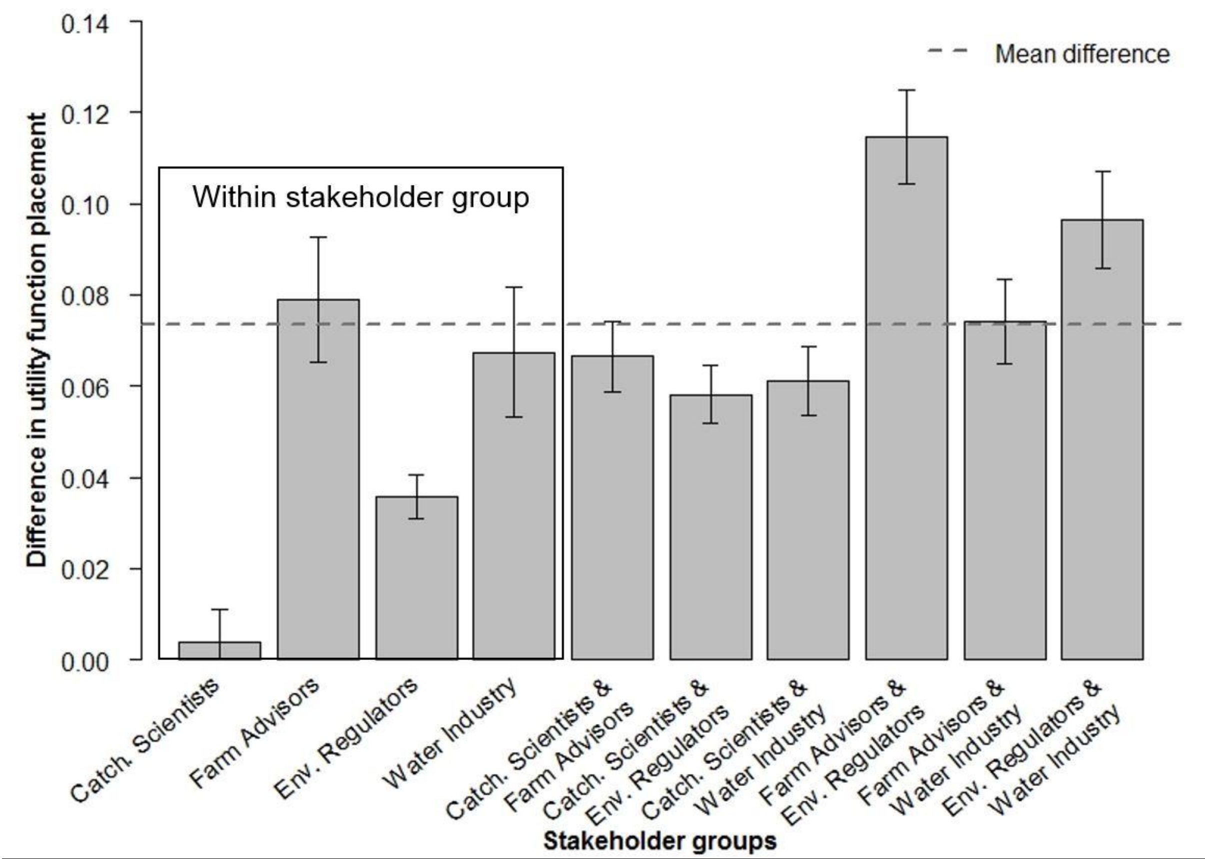


Fig. 3.8: Mean differences between utility function placements by individuals within their own stakeholder group, and between the other stakeholder groups. Error bars indicate ± 1 standard error.

3.4. Discussion

Using a novel mixed-method approach we have identified differences in trade-off prioritisations across the stakeholder groups surveyed, highlighting the importance of including participatory approaches in ecosystem service trade-off analysis. Expert judgement is vital for implementing the ecosystem service concept in practice and making use of existing knowledge and expertise may at times be preferable to collating large amounts of data through ecosystem service assessments (Jacobs *et al.* 2015). Our trade-off analysis was able to elicit robust responses as shown by the consistent rank ordering of the four stakeholder groups in both the self-determined PPF and the threshold PPF, as well as through the consistency in placement of the utility functions by the control group of participants who made a repeat response on the threshold function.

Our methodology provided a rapid and engaging method for assessing stakeholder perceptions, knowledge and preferences of an ecosystem service trade-off relationship while incorporating perceived social demand of the ecosystem service interaction by key stakeholder groups. The results highlighted differences in how stakeholder typologies view PPFs and utility functions in their catchment, indicating potential for conflict between stakeholders and possible barriers to integrated decision-making.

The finding that a number of Farm Advisors did not agree in either of the proposed PPFs is of particular practical relevance for land and water management decision-making and further highlights the lack of a common underpinning understanding among some stakeholder groups and a need for 'engagement as mediation' (Reed *et al.* 2018). While farmers are aware of some of the effects of agriculture on aquatic health, their understanding may be more relevant for their day-to-day activities (Lamarque *et al.* 2011), and may benefit from strengthening their knowledge on how agricultural management affects ecological status of water bodies. Arguably, the agricultural advisors surveyed in our study have a greater understanding of the effects of agricultural intensification on the environment than regular farmers, but still show significantly differing views to other stakeholder groups. Farm advisors with in-depth knowledge of the effects of agricultural management on ecological status could act as intermediaries between environmental regulators and farmers and other farm advisors, since communicators with a shared worldview are more likely to resonate with that particular audience (Kahan *et al*. 2012).

If stakeholders do not agree on the underlying biophysical limits within a trade-off, they are unlikely to reach agreement when it comes to determining how the trade-off should be managed as divergent stakeholder perceptions act as a major barrier to collaboration (Porras *et al.* 2018). Estimating PPFs for contentious trade-offs could therefore provide a mechanism to improve stakeholder understanding of ecosystem functioning. Researchers could play a leading role here as actors to promote stakeholder cooperation and knowledge sharing, aid implementation of innovative land management practice, and advise the farming community on the environmental and socio-economic consequences from unsustainable agricultural practices (Schröter *et al.* 2015). This is supported by our findings that the Catchment Scientists responded not only most similarly within their group but their responses also corresponded closely to the mean from all stakeholders, which may indicate more precise and balanced insights into the socio-ecological system, reflecting their role as outside observers, seeking unbiased, objective descriptions of reality (Rose & Parsons 2015). Catchment Scientists were also the only group not to differ in where their utility function was placed by the other three stakeholder groups, which again perhaps reflects on their impartiality.

At a more theoretical level, the variability observed for the other stakeholder group responses may reflect the challenge of making cross-disciplinary trade-off assessments and the disciplinary nature of expertise partly informing the principle of expert judgements (Fish *et al.* 2009). Catchment Scientists also tended to select large confidence intervals while Environmental Regulators were more likely to select intermediate uncertainty around the mean of the PPF. Arguably, regulators and policy makers are less comfortable with acknowledging higher levels of uncertainty relative to those working in academic fields where communication of uncertainty is considered an important component of reporting results (Morss *et al.* 2005). Ecosystem service trade-off relationships are, however, complex and vary depending on heterogeneous and stochastic biogeophysical processes, but also due to spatial and temporal differences in land use, which introduces uncertainty into trade-off analysis and may have influenced the variability in the confidence intervals reported by our participants (Lu *et al.* 2014).

In our study participants had to estimate the potential impacts of increased agricultural intensity on WFD ecological status for their entire catchments. This contributed a large amount of uncertainty to their judgement, which is likely why we did not see any differences between catchments. This may be addressed in future studies, however, by estimating PPFs within a study catchment using spatially explicit models such as InVEST (Integrate Valuation of Ecosystem Services and Trade-offs) or SWAT (Soil and Water Assessment Tool; Cord *et al.* 2017). Given that measures we used in our application of the methodology were relatively broad and incorporated a number of ecosystem services, differences in stakeholder perception of these may have influenced the results as well. When interpreting the results it is important to remember that the stakeholder responses incorporated their cultural values, as well as their perception of the socio-economics of the trade-off and their views on the institutional specificities of their own and the other stakeholder groups. Incorporating expert judgements can deliver benefits to ecosystem service assessments; however, it may be difficult to disentangle such perceived judgements from the underlying socio-ecological processes. Although expert judgements are more liable to biases than other techniques due to tendencies such as overconfidence and anchoring (Mach *et al.* 2017), they may also assess trade-offs and uncertainties in ways that are not otherwise possible and can provide logical arguments to support their judgements (Singh *et al.* 2017). Expert knowledge may also provide time-integrated assessments, as opposed to momentary snapshots and can interpolate or extrapolate when ecosystem services may not be measured directly (i.e. Martin *et al.* 2012). Making use of a 'thought experiment', such as that used in our methodology, can extract stakeholder experience and acquired instinct to capture estimations which could not have been measured in the field.

There were also clear differences between Farm Advisors and Environmental Regulators in estimating utility functions. Farm Advisors scored utility functions toward lower agricultural intensity for their own, together with the other typologies; whereas the Environmental Regulators perceived all stakeholder groups to prefer higher agricultural intensity than the mean results suggested. Given the natural potential of these two groups for conflict due to their competing priorities, this misconception, or lack of understanding of the opposing group's interests may further exacerbate tensions (Petersen-Perlman *et al.* 2017). These differences are likely due to the nature of their professions, for example, environmental regulators are driven by EU legislation to avoid declines in ecological status of water bodies, while a priority for farm advisors is often the financial viability of agricultural systems. This is an important point because respondents were asked to participate as professionals and not as individuals, though it is difficult to ascertain whether personal preference could ultimately influence their choice (Nordén *et al.* 2017). This is particularly true when ecosystem service interactions are antagonistic, which might lead to tensions and inconsistencies in professional judgements and personal views (Barnaud *et al.* 2018).

If land management policies continue to increasingly focus on providing multiple ecosystem services, farmers may end up as the main 'losers' due to reduced provisioning services, exacerbating conflicts between farmers and regulators (Kovács *et al.* 2015). Adapting the approach used in one-to-one interviews here for the context of a group discussion may therefore present an opportunity for stakeholders to articulate their utility functions and allow different organisations to improve their mutual understanding of each other's priorities and conflicting goals in a non-confrontational and abstract setting (Cebrián-Piqueras *et al.* 2017). Reducing bias in how stakeholders view their catchments could positively affect the capability of people to cooperate effectively and may, in turn, help to highlight 'win-win' opportunities in land and water management (Vallet *et al.* 2018). Although unprompted, when discussing PPFs and utility functions at the start of the exercise, a number of Farm Advisors, Environmental Regulators and Catchment Scientists mentioned that their work aims to change the shape of the PPF in their catchment to allow for higher agricultural intensity without compromising ecological status. The difference in the placement of utility functions on the threshold PPF illustrates this as utility functions shifted towards higher agricultural intensity without compromising ecological status. This presents a potential win-win opportunity, particularly between Farm Advisors and Environmental Managers to improve their utility functions by shifting the PPF through land-based management techniques, such as expansion of riparian buffer zones and agro-forestry, and increased production of legumes (Howe *et al.* 2014).

Arguably, the shape of the PPF can help determine how a trade-off should be managed, with more fragile relationships, such as an exponential decline pointing towards land sparing, while a more resilient relationship may allow more land sharing (Maskell *et al.* 2013). If a catchment is able to sustain greater agricultural intensity without compromising ecological status of its water bodies, it may be more resilient, i.e. due to deep soils buffering agricultural inputs. The tendency of Farm Advisors to select the threshold PPF and for a number of them to disagree that increased agricultural intensity decreases ecological status, indicates that they believe their catchments to be relatively resilient and able to sustain larger amounts of agriculture without impacting ecological status, or even having a positive effect on it. This contrasted with Environmental Regulators who more frequently identified with the logistical decay function, which represents a more fragile relationship between the two services, and may imply that larger areas of the catchment should be given over to land-sparing and mitigation measures to ensure good ecological status.

The ease of application and simplicity of our methodology make it a promising approach for embedding stakeholder views into ecosystem service trade-off analysis. This is important because even though the recognition of the nuances and complexities of ecosystem service trade-offs has improved, quantitative evidence and an accurate characterisation of how ecosystem service interactions manifest is needed to ensure sustainable management of ecosystems and to maximise the benefits they provide to humans (Spake *et al.* 2017). Our approach also has generic transferability to allow for the capture of views from other users, such as local residents or tourists, as these stakeholders are often the most impacted by ecosystem service trade-offs (Turkelboom *et al.* 2018). This may be especially useful in assessing the impacts of potential management options on cultural ecosystem services, such as landscape aesthetics, which are inherently difficult to estimate.

The flexibility of this method means it may easily be applied to elicit stakeholder views on how an ecosystem reacts to other land use changes, environmental pressures, or more specific ecosystem services, such as increases in tree cover or point source pollution. Although our approach is limited by only assessing the trade-off between two ecosystem services, future application of it could include multiple conflicting objectives. The methodology could also be used in conjunction with catchment modelling software to find optimum levels for certain ecosystem service provisioning, or with multi-objective programming to include PPFs of a number of trade-offs (e.g. Groot *et al*. 2018). Spatio-temporal simulation models such as InVEST (Han *et al.* 2017), ARIES (ARtificial Intelligence for Ecosystem Services; Villa *et al.* 2014), or SWAT (Francesconi *et al.* 2016) are often used to model ecosystem service trade-offs and their coupling to participatory research to help moderate outputs may provide a useful avenue for future research. We consider that this methodology could potentially be incorporated into awareness-raising programmes in catchments as part of a participatory approach to engage stakeholders. In doing so it could promote discussion of otherwise implicit decision-making, build shared mutual understanding to facilitate future cooperation, or assess whether stakeholders could be offered compensatory payments for utility losses (Brunet *et al.* 2018; King *et al.* 2015). The ease of use of the methodology could also allow for longitudinal analysis of how stakeholder perceptions change over time, which is an aspect of integrated catchment management that we know very little about(Stosch *et al.* 2017). Finally, allowing stakeholders to score utility functions on PPF curves offers a solution to integrating social demand into trade-off assessments, which often defy measurement and are hence widely underrepresented (Satz *et al.* 2013).

3.5. Conclusion

This study shows the importance of participatory trade-off analysis due to the differences in how stakeholders prioritise trade-off preferences arising from ecosystem service interactions. Valuing stakeholder knowledge as a form of expert data and integrating this into participatory decision-making processes for land and water management thus contributes considerable value beyond traditional approaches to ecosystem service assessments. Our results suggest that to achieve sustainable management of socio-ecological systems it is insufficient to focus on optimising ecosystem service trade-offs alone, as this fails to capture the social dimensions associated with end-user interactions when balancing the often competing demands of different stakeholder groups. Using participatory trade-off analysis can therefore reveal potential sources of conflict and/or synergies among stakeholder groups. In turn, approaches like this can support interdisciplinary research to better our understanding of the socio-ecological complexity of catchment systems and the management of ecosystem service interactions to deliver multiple benefits for stakeholders with differing environmental management remits.

4. Catchment-scale participatory mapping identifies stakeholder perceptions of land and water management conflicts

4.1. Introduction

Sustainable management of natural resources is under pressure from environmental and social drivers, such as climate change and global population growth (Ye *et al.* 2020; Khan *et al.* 2021). If landscape-scale decisions are implemented in response to environmental and social drivers without full consideration of the socio-ecological complexity of catchments, then there is the potential for unintended consequences for both the environment and society. For example, a loss of biodiversity or heightened risk to human health and well-being (Shepherd *et al.* 2016; Schmeller *et al.* 2020). Therefore, integrated catchment management is often described as a 'wicked problem' because solutions can be difficult to identify and implement due to the uncertainty, complexity and divergency of stakeholder interests (Rittel & Webber 1973; Kirschke *et al.* 2018). The potential for increased land use conflict, arising when stakeholders have incompatible interests concerning land use and resource management (Von Der Dunk *et al.* 2011), becomes more likely as demands and pressures on catchments and coastal waters grow or become more diverse (Durance *et al.* 2016; Mendenhall *et al.* 2020). Thus, identifying sustainable solutions to wicked problems requires adaptive and integrated decision-making in land and water management (Pahl-Wostl et al. 2008; Vasslides & Jensen 2016; Mohamad Ibrahim et al. 2019), and highlights the importance of stakeholder-focused approaches (Mason *et al.* 2018).

Land use conflict is frequently driven by rapid land use change (Abram *et al.* 2017), resource scarcity (particularly water scarcity) (Adams *et al.* 2019; Hummel 2017), or power imbalances (Boelens 2014) and can increase social tensions, political instability and, in the most extreme cases, can lead to violent conflict (Cusack *et al.* 2021; Eliasson 2015). In more stable, affluent societies, the causes of land use conflict are often due to changes in land management affecting broader, often difficult to measure, socio-ecological aspects, such as noise, odour, negative visual impacts, or risks to biodiversity conservation (Klæboe & Sundfør 2016; Young *et al.* 2005). Trade-offs are often expressed downstream, in lower part of the catchment (Asquith *et al.* 2008; Stosch *et al.* 2017), and can cause severe disadvantages to affected communities or even entire national economies (Munia *et al.* 2016). Due to hydrological connectivity, river catchments often integrate multiple pressures, e.g. from point and diffuse source pollutants, abstraction and physical alterations of the landscape (Elosegi & Sabater 2013; Heathwaite 2010). This puts riparian ecosystems and their critical natural resources at risk (Albert *et al.* 2021), and reinforces the concept of the 'catchment' being the most appropriate scale to holistically manage land and water environments (Johnson *et al.* 2017).

Ecosystem service mapping techniques can identify likely trade-offs between different land management decisions in catchments (i.e. Lautenbach et al. 2013; Karabulut et al. 2016), or model estimates of land use conflict, i.e. using spatial indicators of the potential for soil erosion and the quality of agricultural land (Kim & Arnhold, 2018); however, there are many factors that mapping efforts cannot take into consideration, especially socio-cultural benefits and dis-benefits (Harrison *et al.* 2010). Trade-offs between ecosystem services manifest themselves in complex interactions (Pilgrim *et al.* 2010) and bundles of services respond to changes in a variety of social, environmental, economic and political drivers across both space and time (Rodríguez *et al.* 2006; Spake *et al.* 2017). Accurately modelling and mapping ecosystem services in catchments is therefore challenging, due to the stochastic and often non-linear and even chaotic nature of socio-ecological systems (Spears *et al.* 2012; Walker *et al.* 2003). Furthermore, ecosystem service mapping studies often lack sufficient primary, scale-appropriate data, and frequently fail to validate their modelled results or assess uncertainty in their models (Martínez-Harms & Balvanera 2012; Seppelt *et al.* 2011).

These shortcomings suggest a role for including the knowledge and perspectives of experts, stakeholders and local communities in assessing the potential for ecosystem service trade-offs and land use conflict in catchments. Widening participant involvement in research can identify differences in opinion and understanding across different stakeholder communities, capture valuable local expertise and understanding, and facilitate knowledge exchange among stakeholders (Darvill & Lindo 2016; Galafassi *et al.* 2017). Participatory research approaches can also ensure that management decisions are more inclusive and socially acceptable, and are implemented more effectively (Etienne *et al.* 2011; Oliver *et al.* 2017). Participatory mapping, where spatial information is used or produced as part of a participatory process, is a useful methodology to support the integration of knowledge from multiple contributors and to elicit information which may be challenging to estimate using only quantitative data. Participatory mapping studies are often used to understand how stakeholders value ecosystem services (Brown & Fagerholm 2014; Klain & Chan 2012; Mahboubi *et al.* 2015; Plieninger *et al.* 2019; Zoderer *et al.* 2019), or to map, understand and mediate conflict (Cronkleton *et al*. 2010; Brown & Raymond 2014; Philpot *et al.* 2019). Participatory mapping has also been used to inform mitigation measures by identifying spatial hotspots of ecosystem service benefits and areas at risk of competing demands (Bryan *et al.* 2010; Brown & Raymond 2014; Reilly *et al.* 2018). To date, relatively few participatory mapping studies have used both quantitative and qualitative data to complement and validate their methodologies (Brown *et al.* 2017).

In this study the perceptions of four distinct land and water management stakeholder groups were identified in three contrasting catchments by applying a novel, mixed method, stakeholder engagement methodology that incorporates participatory conflict mapping with a qualitative survey. The objectives were to: (1) identify and characterise spatial hotspots of perceived conflict in catchments of varying land use and environmental characteristics, (2) quantify how perceived conflicts differ among key groups of stakeholders within each catchment, (3) model whether land cover data and other variables can be used to predict conflict and land use competition in the study catchments, and (4) use supporting qualitative data to interrogate the potential drivers of any perceived land use conflicts and likely solutions to the issues.

4.2. Methodology

This study developed a mixed method participatory mapping exercise coupled with a qualitative survey as part of a novel stakeholder engagement methodology. Combining quantitative spatial data and qualitative responses allowed us to determine how participants from four key stakeholder groups perceived conflict and competing demands on water resources (herein termed conflict) within their local catchments, as well as the underlying drivers of conflict and any perceived likely future changes to such demands. Three contrasting catchments with diverse geomorphologies, land cover composition, stakeholder communities and land and water management pressures were selected from across Scotland, UK (Fig. 4.1).

4.2.1. Study catchments

The River Spey drains an area of just under 3000 km^2 in the north-east of Scotland and lies within the north-western part of the Cairngorms National Park. Land cover is dominated by moors and heathland (29%), sparsely vegetated highlands (23%) and coniferous forest (16%; Table 4.1). This catchment has a varied land use including rough grazing, commercial forestry, arable farming and sporting estates. Whisky distilleries, tourism and angling also represent important local industries, with 8% of Scotland's total wild salmon catch originating from the river Spey. The river is a Special Area of Conservation (SAC) for Atlantic salmon, sea lamprey, freshwater pearl mussel and otter, and forms part of the EU Natura 2000 network (Joint Nature Conservation Committee 2013). An estimated 20% of the mean annual water flow to Spey Bay is diverted to large hydropower schemes in nearby catchments. Competing demands on the remaining water resources in the catchment are local hydropower plants, a growing food and drink manufacturing industry, and increasing domestic water demands and irrigation needs (Fleming & MacDougall 2008).

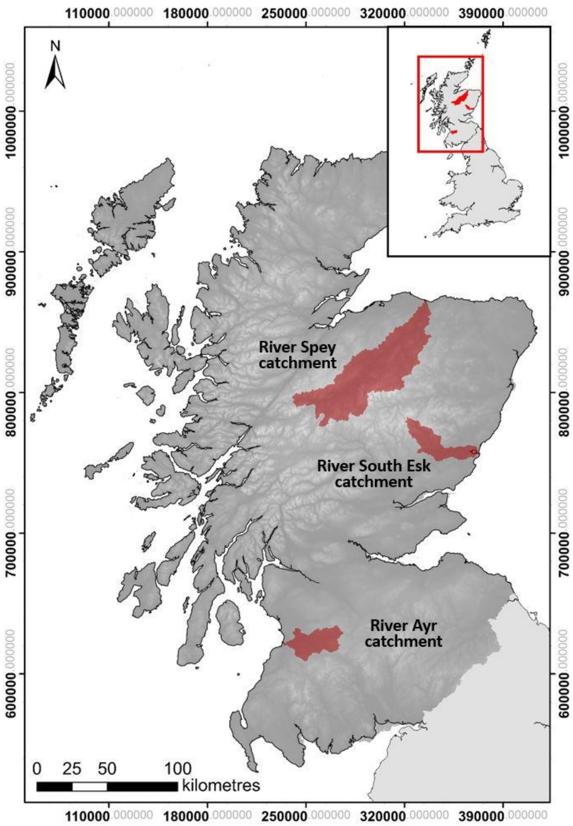


Figure 4.1: Location of the three study catchments in Scotland, UK.

In the east of Scotland, the South Esk catchment has an area of just over 600 km^2, originating in Glen Clova, in the Cairngorms National Park and draining into Montrose Bay. The geomorphology and land cover are distinctly split between its upper catchment at higher altitudes, which is dominated by moors and heathland (33 %) used for rough grazing, and its more gently sloping lower catchment, which is dominated by arable land (31 %) and improved grazing (12 %). The catchment has been designated as a SAC for Atlantic salmon, freshwater pearl mussel and otter. It has also been identified as a Nitrate Vulnerable Zone as both ground and surface waters are at risk of nitrate leaching from soils; therefore, farmers in the catchment must adhere to additional management restrictions to reduce nitrate leaching in the catchment. Point source pollution from wastewater effluent discharge, diffuse pollution from agriculture, and water abstraction for arable farming are the major pressures on this catchment system (South Esk Catchment Partnership Steering Group 2009).

Table 4.1: Land cover types in the three study catchments as a percentage of overall area covered.

Land cover type	(% land cover) Spey	Esk	Ayr
Moors and heathland	29.07	32.91	11.47
Coniferous forest	15.89	8.45	9.38
Pastures	9.18	12.09	39.46
Sparsely vegetated areas	22.61	0.03	0.00
Natural grasslands	8.54	9.81	14.39
Arable land	2.01	31.20	7.04
Peat bogs	6.57	0.86	10.11
Transitional woodland-shrub	2.71	1.34	2.02
Broad-leaved forest	1.95	1.41	1.15
Urban areas	0.52	1.03	2.10

The River Ayr catchment drains an area of just under 600 km^2 into the Firth of Clyde. Approximately 65% of the catchment area is pasture with lowland improved grassland used for intensive dairy farming and upland pastures supporting sheep and beef farming; diffuse pollution from agriculture represents a significant challenge for water quality and human health, the latter being especially important due to a number of designated public bathing water beaches on the Ayrshire coast. Livestock rearing, tourism and wild salmon angling are important local economies, which impose contrasting demands on the catchment. The Scottish Environment Protection Agency (SEPA) has declared the catchment a 'priority catchment for diffuse pollution' and has worked with local farmers to avoid breaches of local regulation, which have primarily been caused by livestock erosion of riverbanks and their direct access to watercourses.

4.2.2. Sample selection and engagement design

In each of the three study catchments, three to five individuals from four key stakeholder groups took part in the exercise: Environmental Regulator Staff (n = 12; all from SEPA), Water Regulator Staff (n = 9; all from Scottish Water, Scotland's public water and wastewater company), Catchment Scientists (n = 11; from universities and research institutes across Scotland) and Farm Advisors (n = 11; from the National Farmers Union Scotland as well as independent farm consultants). A total of 43 stakeholders carried out the survey in 2017, 15 of which had local expert knowledge on the River

Spey catchment, 13 on the South Esk and 15 on the Ayr. The full details of the stakeholder sample selection can be found in (Stosch *et al.* (2019) but in summary this was based on stakeholder knowledge of the catchment and their expertise in land and water management issues. Participants were initially identified through a desktop study with additional stakeholders identified through recommendations from the initial cohort.

The participatory mapping exercise and the interview were conducted face-to-face to ensure that the tasks and their context were understood and to avoid any miscommunication or misinterpretation between the interviewer and participant. All interviews were conducted by the same interviewer to ensure consistency in approach and the results anonymised so that the stakeholder group of each respondent was known, but not their identity. The interviews were carried out on a one-to-one basis to allow stakeholders to express their perspectives frankly, even if their views deviated from the majority-held opinions within their stakeholder group. The interviews were developed to elicit responses to 12 questions, prompting participants to consider their views about the major themes of ecosystem service provisioning and trade-offs, competing land use, and conflict among stakeholder groups in their catchments (see Appendix 1 for the full set of survey questions). Of central importance was Question 8 whereby participants were asked to spatially identify areas on a catchment map where they perceived there to be conflict among stakeholder groups. As there are no 'severe' water or land use conflicts in Scotland, a couple of participants in the first few interviews thought the word 'conflict' was too strong, so the definition in the survey was broadened to include areas of competition between land uses as a proxy of a driver of conflict (Jensen *et al.* 2019). Participants were asked to delineate around the areas of conflict/competition that they identified on a map of their catchment, and they were also told that they could select the entire area of a particular land cover type. Hardcopy maps were used for this exercise, as opposed to digital mapping, as this proved more accessible to participants during pilot interviews, with both formats shown to produce equivalent mapped outputs (Pocewicz *et al.* 2012). Following the mapping exercise, the remaining questions of the interview focused on collecting qualitative evidence to support why particular areas had been selected and to identify the possible drivers behind land use competition and how they may be resolved or may change in the future. The interviewer recorded notes of the participants' responses to all of the questions. The interviews were also audio recorded and transcribed, however, only the interviewer's notes were used for the analysis presented here.

4.2.3. Analysis

Responses from the participatory mapping exercise were digitised from the scanned paper maps to polygons in ArcMap 10.4.1 (Esri 2016). The polygons were overlaid in ArcMap to generate heat maps of conflict for each catchment, where higher numbers of overlap in stakeholder perception equate to increased 'heat' of conflicts and land use competition. To quantify where each of the polygons from different participants overlapped, the polygons were first turned to raster files and resampled to 100 m^2 resolution as well as 'snapped to raster' in their processing extent to make sure all the files were aligned. To identify the overlapping raster squares between different respondents, the polygons were reclassified using numbers as place holders for each participant. Numbers which were powers of two were used, or 2^n (i.e. 1, 2, 4, 8, 16...), as they could then be overlapped and summed in ArcMap prior to extraction as an Excel file while retaining the identifying information of each stakeholder. To compare the amount of overlap between different stakeholder groups, the number of overlapping raster squares per stakeholder group were normalised by subtracting the mean number of rasters per participant from their stakeholder typology and dividing by the standard deviation, which accounted for the variation in raster square selection between the stakeholder groups and the differing scales of the catchment. The differences in overlap of participant's highlighted area of conflict between the stakeholder groups and between catchments were compared using a Kruskall-Wallis statistic.

Regression analysis was used to model whether land cover data and other variables such as stakeholder group or catchment type could be used to predict conflict and land use competition in the study catchments. The raster data from all of the heat maps were combined with 2015 CEH landcover data in ArcMap and extracted to R using the 'tiff' (Urbanek 2003), 'raster'(Hijmans 2015) , 'sp' (Pebesma & Bivand 2005) and 'rgdal' packages (Bivand *et al.* n.d.). The 'brick' function was used to extract the raster information from ArcMap into R and combined data from the three catchments into one file. For the regression analysis, general linear models were used to determine how perceived conflict may be explained in part by land cover type, catchment or stakeholder group identity. The final model was selected as all factors were highly statistically significant ($P < 0.001$) and its Akaike Information Criterion was lower than other more and less complex models.

The responses from the survey were grouped into themes according to content. The themes were: types of ecosystem services, types of conflict and types of drivers. This categorisation allowed comparison between groups and catchments. The responses to Question 9 ("*What are the major drivers for conflict between land use, ecosystem service provision and stakeholders in your catchment?*" were initially grouped into 16 categories, with the nine smallest categories being grouped together as 'Other', resulting in eight groups: 'Policy', 'Subsidies', 'Climate change', 'Financial pressures', 'Competing interests', 'Urban population growth and increased tourism', 'Lack of communication and integrated management', and 'Other'.

4.3. Results

Participants identified a total of 97 areas of conflict within the participatory conflict mapping exercise. The most polygons were mapped in the Spey catchment (41), followed by the Ayr (34) and the Esk (22). Catchment Scientists were the stakeholder group that identified most polygons (33), followed by Environmental Regulator Staff (27), Farm Advisors (24) and Water Regulator Staff (13).

4.3.1. Hotspots of conflict

The heat maps of perceived conflict, combining all the stakeholder's responses, showed that perception of conflict in all three catchments was often congruent among participants, with up to ten stakeholders identifying the same localised hotspots in their catchment. Participants also identified more dispersed issues across the catchments, which were manifest at a landscape scale

(Fig. 4.2). In the River Spey and South Esk catchments the most commonly identified areas were more localised. Stakeholders in the Ayr catchment identified areas of conflict more widely across the catchment, and the mean proportion of perceived conflict across the catchment was larger with 22.44% ± 0.04 followed by the Spey (18.71% ± 0.02) and the South Esk (17.17% ± 0.04). Results from the survey data also suggested that there were more diverse issues in the Ayr catchment compared to the other two catchments.

4.3.1.1. The Spey catchment

In the Spey catchment, two clear hotspots emerged from the participatory mapping. The more northerly hotspot (which is further downstream) is situated around a town popular with tourists, where stakeholders asserted that this holiday destination has experienced an increase in urban development and tourism influx in recent years. The hotspot, with an area of around 9 km^2, was identified by ten out of the fifteen Spey stakeholders. Development in and around the town was thought to cause significant socio-ecological pressures, such as increasing house prices and demands on the local water resources for potable use, in addition to increasing abstraction pressures on the Spey. Further upstream, the second hotspot (of about 13 km^2 and identified by nine participants) was perceived to be where a significant proportion of the river's water was being diverted to a neighbouring catchment to power a hydroelectric dam. The upstream abstraction of water to generate energy in a neighbouring catchment was widely mentioned by participants in their survey responses as causing compromised downstream flows, and concern for aquatic biodiversity (especially for conservationists, anglers and river trusts). The third most commonly identified area (7 out of 15 stakeholders) of conflict in the Spey was due to increases in tourism and recreation in the uplands around the local town, which is perceived to cause conflict with anglers, sporting estates and biodiversity conservation efforts. Five participants mentioned that an increased use of the area was causing conflict with biodiversity conservation; however, four stakeholders also mentioned that conflict situations were arising among different groups of recreational users, such as between anglers and wild water rafters or between shooting estates and hill walkers.

4.3.1.2. The South Esk catchment

In the South Esk catchment the most intense perceived conflict around a lowland town was identified by ten out of the thirteen stakeholders. The complementary survey determined that a recently installed large flood control scheme had caused extreme flooding and erosion of farmland downstream of the town. Out of the ten stakeholders that identified conflict within this 8 km^2 area, seven specifically identified the town as an area of conflict. Throughout the rest of the catchment, overlaps were much less common although more extensive in area, with a maximum of six overlaps near the urban and coastal areas. Seven stakeholders from the South Esk and the two other catchments also voiced concerns over the Scottish Government's target to plant a minimum of 10 000 ha of forestry per year to offset greenhouse gas emissions. They were concerned about this encroaching on the availability of upland grazing spaces, as trees would not be planted on high quality agricultural land nor on peatlands which would likely emit greenhouse gases when forested. Other trade-offs mentioned were the impacts of future tree felling on downstream water quality, and how wet woodland planting may impact wading bird conservation as it would provide cover for predators such as birds of prey.

4.3.1.3. The Ayr catchment

Responses of up to eight stakeholders overlapped in the Ayr catchment; however, in contrast to the other two catchments, these did not form spatially distinct or acute hotspots. Although conflict was commonly identified around distinct locations in the Spey and South Esk, stakeholders from the Ayr catchment perceived conflict more widely across the catchment with larger areas showing overlaps of five or more stakeholders (18% of total area). The areas of conflict identified by five or more stakeholders in the Spey catchment were only 5%, and only 3% in the South Esk catchment. The greatest number of overlaps in the Ayr catchment were situated around the urban and coastal areas of Ayr, which is the main town in the catchment (eight out of 15 stakeholders), in the upland areas of the catchment (seven stakeholders), and within improved grassland areas immediately surrounding the River Ayr and its tributaries (six stakeholders).

The qualitative data from the survey showed that although the numbers of conflict and drivers raised by stakeholders in the Ayr were comparable to those in the other two catchments, they revealed a more diverse array of issues. In terms of the area of conflict identified around the town of Ayr, seven out of fifteen stakeholders mentioned concerns around bathing water quality being impacted by farming and particularly poaching of cattle. Five stakeholders perceived urban development to cause a loss of prime agricultural land and flooding. And seven stakeholders voiced concern that increases in forestry in the catchment is leading to farmers becoming frustrated by rising land prices for rough grazing, which was being exacerbated by the Scottish Government's 10000 ha tree planting targets. Four stakeholders also mentioned that mining areas that have been historically poorly managed are now left unused and unrestored and three of the fifteen Ayr stakeholders thought there was conflict due to visual impacts of increasing windfarm development in the uplands.

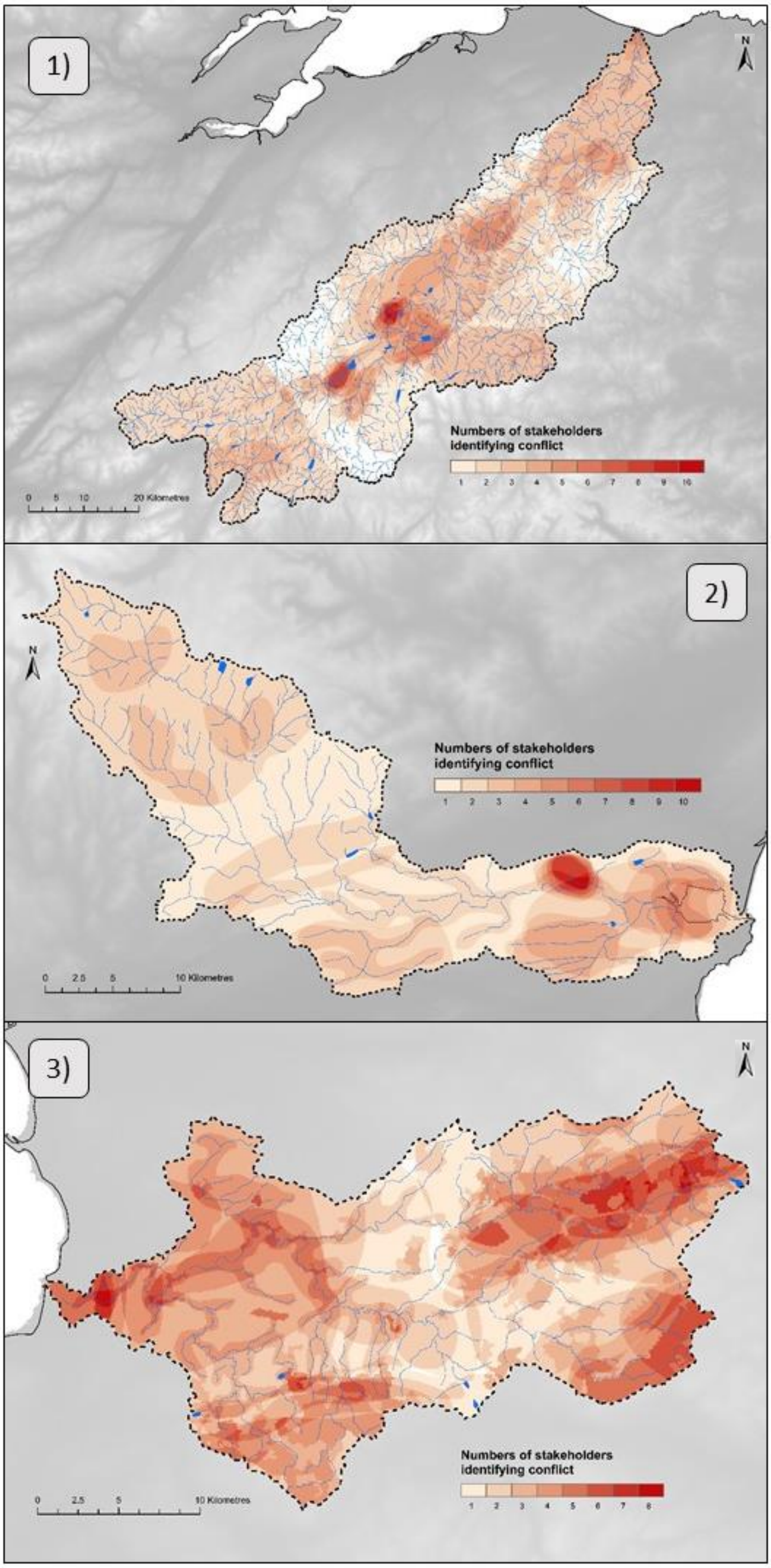


Figure 4.2: Heat maps of perceived conflict in the River Spey (1), River South Esk (2) and River Ayr catchment (3).

4.3.2. Stakeholder typologies and conflict

In the Ayr catchment, Water Regulator Staff selected fewer, and smaller, areas of land use competition in the participatory mapping exercise and responses within this stakeholder group only overlapped in the very west of the catchment around the town of Ayr (Fig. 4.3). The other stakeholder group's responses cover larger areas of the catchment and overlap up to a maximum of three times for Environmental Regulator Staff and Catchment Scientists and up to four times for Farm Advisors. Water Regulator Staff identified only five polygons of perceived conflict in the Ayr catchment, with a total of 89.9 km^2, whereas Environmental Regulator Staff identified 11 polygons with a total area of 873.9 km^2, Catchment Scientists identified seven polygons at 396.4 km^2 and Farm advisors selected 11 polygons and the largest area with 1 008.2 km^2. This trend was similar in the other two catchments, where Water Regulatory Staff also identified fewer, and smaller, areas of potential conflict compared to the other three stakeholder groups.

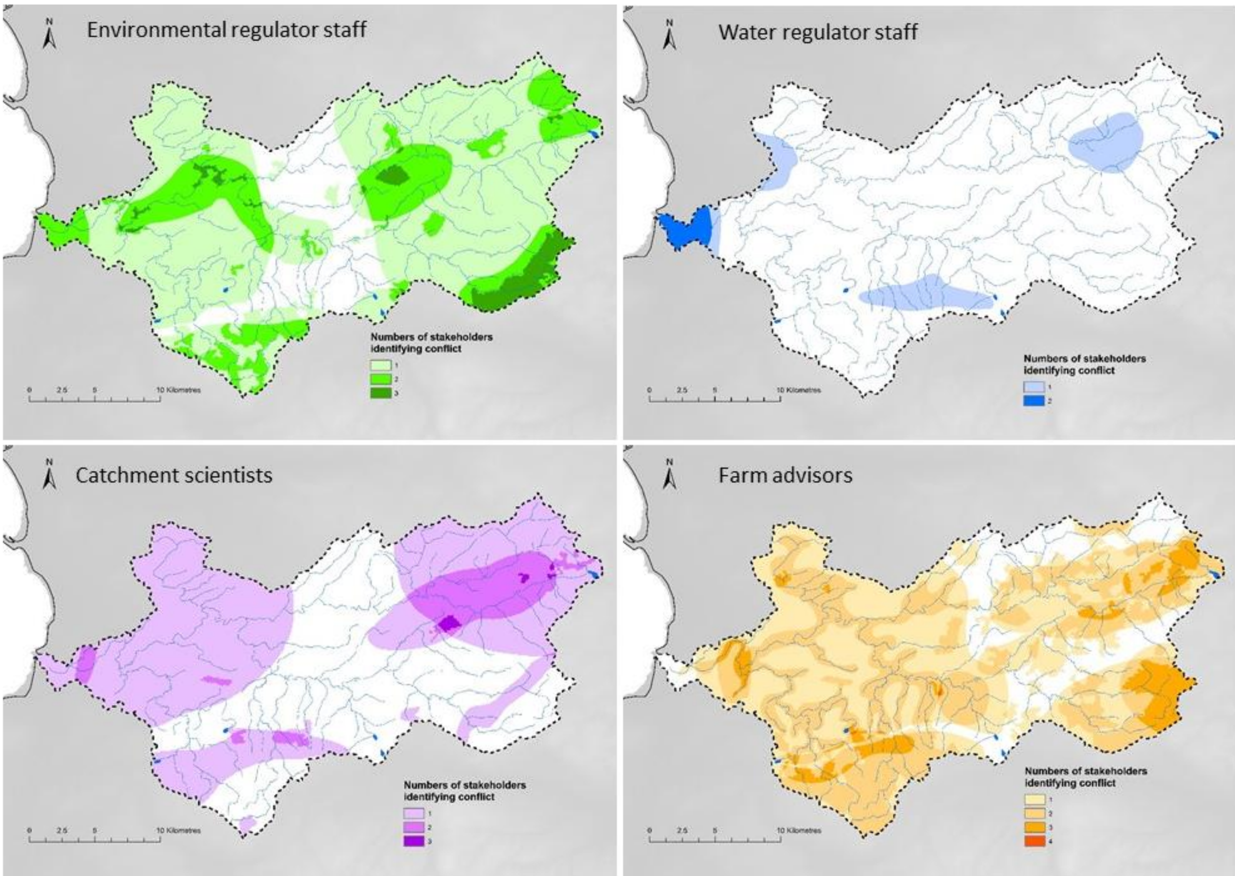


Figure 4.3: Heat maps of perceived conflict by Environmental Regulator Staff, Water Regulator Staff, Catchment Scientists and Farm Advisors in the River Ayr catchment.

4.3.3. Overlap analysis

Overlap analysis revealed that although there was a lower number of normalised overlaps in the South Esk catchment there was no significant difference in the amount of overlap between the catchments ($H = 0.37$, $P > 0.05$), although there was a significant difference among stakeholder groups ($H = 11.6$, $P < 0.01$). The responses of Catchment Scientists were very similar to the mean overlap of responses (Fig. 4.4). Responses from Farm Advisors and Environmental Regulators were on average more similar within their group; however, this was not statistically significant due to the large variation in overlap among stakeholders. Farm Advisors in the Ayr catchment and all Water Regulator Staff showed significantly less overlap within their stakeholder group.

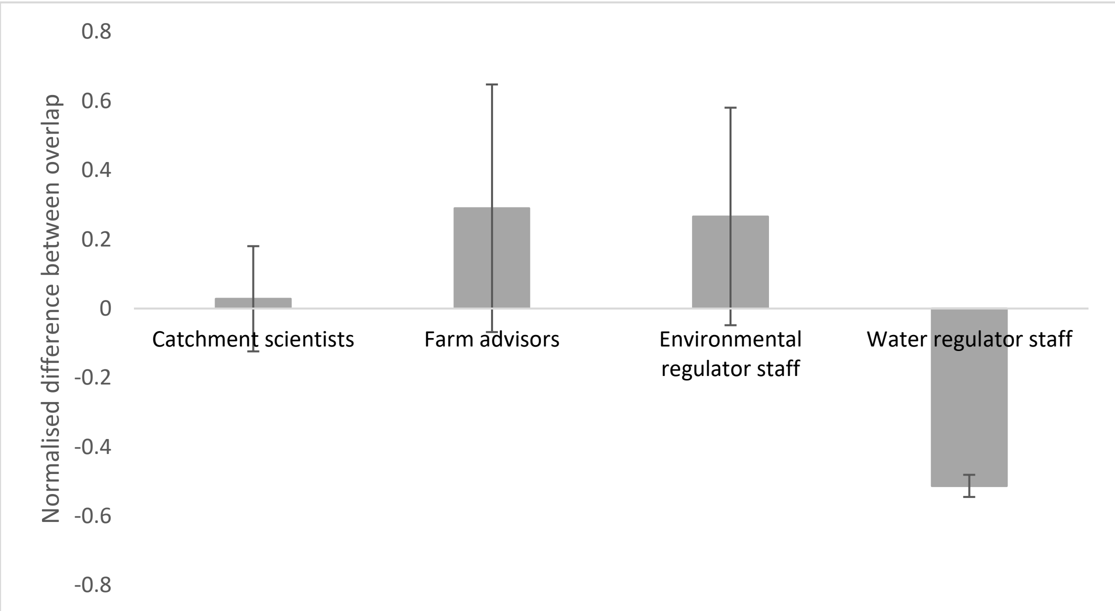


Figure 4.4: Difference in the overlap between stakeholder groups' responses in the participatory conflict mapping exercise, normalised by the mean overlap and standard deviation (± standard error).

4.3.4. Land cover and conflict

The proportion of stakeholders perceiving conflict could be explained in part by land cover, catchment and stakeholder group identity (full model outputs in Table 4.2). The land cover types that were the best predictors of areas with greater perceived conflict in the model were improved grassland and urban areas (Fig. 4.5). Coastal areas had an even larger effect; however, they had much greater variability associated with them compared to all other land cover types, likely due to their smaller size.

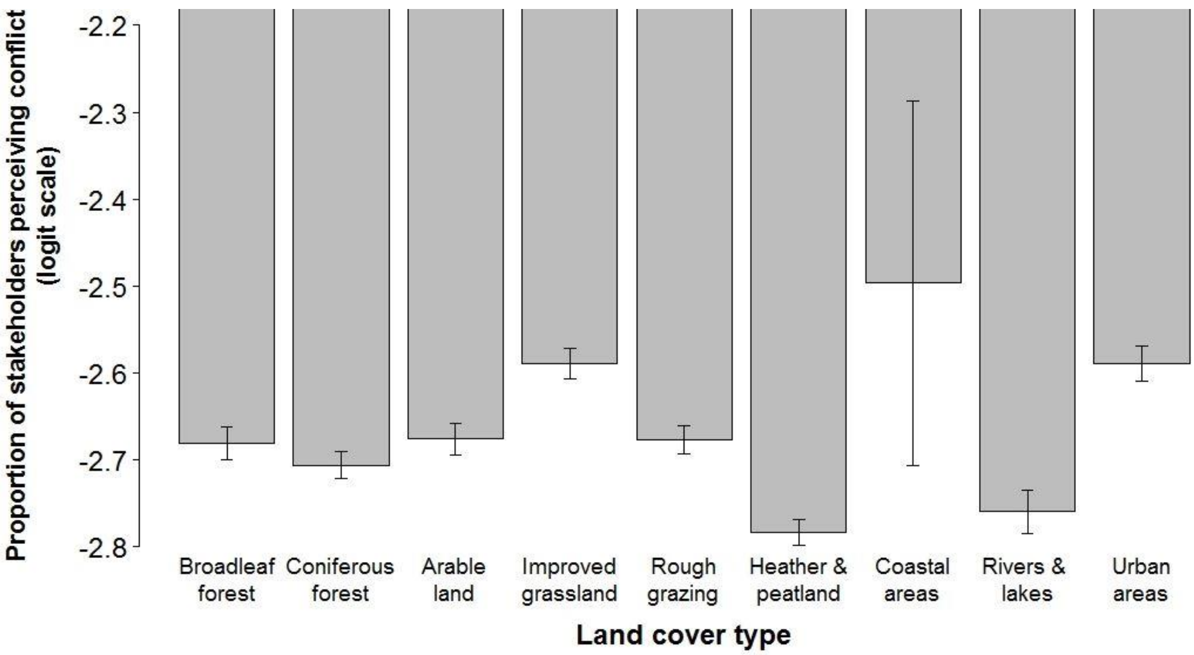


Figure 4.5: Difference between the coefficients of the land cover types in the final model, modelled for the South Esk catchment (± standard error).

With respect to individual catchments, Ayr was associated with greater perceived conflict than the Spey and the South Esk catchments. The modelled proportion of perceived conflict was greatest in the Ayr, followed by the Spey and with the Esk being the lowest.

Results of the modelling for the four stakeholder typologies showed that Farm Advisors were most likely to identify conflict in areas where there was a large proportion of stakeholders that perceived conflict, followed by Water Regulator Staff (Fig. 4.6). Catchment Scientists and Environmental Regulator Staff were the least likely to identify conflict in areas where there was a large proportion of stakeholders that perceived conflict.

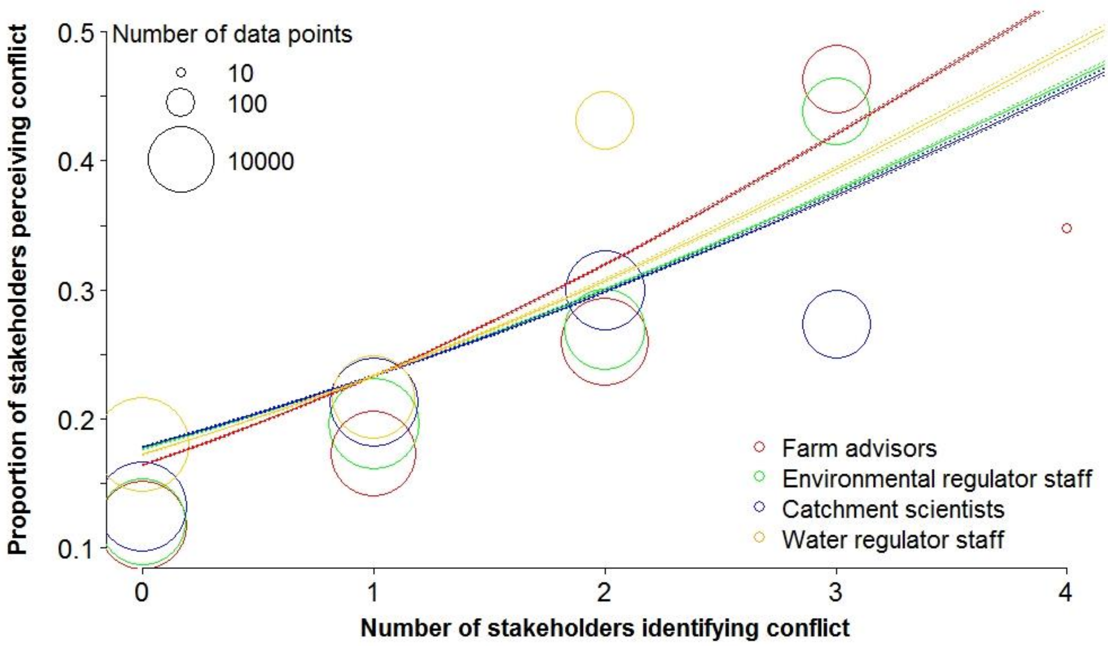


Figure 4.6: General linear model predictions of the relationship between the number of stakeholders from each typology and the proportion of stakeholders perceiving conflict across the catchment (modelling for the South Esk catchment and the broadleaf forest land cover type as they were the first factors and catchment type and land cover were not considered in this model). Data points are displayed using a logarithmic bubble graph to portray the large number of 100 x 100 m rasters used for the analysis.

4.3.5. Drivers of conflict

When stakeholders were asked to comment on the drivers behind conflicts in their catchment, legislation was most commonly mentioned (by 21 of the total 43 stakeholders), followed by financial pressures (13 stakeholders), competing stakeholder interests (11 stakeholders) and urban population growth and increases in tourism (7 stakeholders). These drivers were mentioned by all four stakeholder typologies. Other drivers mentioned by multiple stakeholders were subsidy payments (6 out of 43 stakeholders), climate change (4 stakeholders) and a lack of integrated management and stakeholder understanding (3 stakeholders).

Table 2: GLM binomial modelling outputs of how the proportion of perceived conflict may be explained by land cover type, catchment or stakeholder group.

	Estimate	Std. Error	z value	p-value
Intercept (Broadleaf forest and the South Esk catchment)	-2.68086	0.004817	-556.571	< 0.001
Coniferous forest	-0.02543	0.003976	-6.397	< 0.001
Arable farmland	0.004865	0.004663	1.043	0.296772
Improved grazing	0.091555	0.004476	20.456	< 0.001
Rough grazing	0.004407	0.004119	1.07	0.284673
Upland (moorland and heather)	-0.10336	0.00378	-27.346	< 0.001
Coastal areas	0.183985	0.053147	3.462	< 0.001
Freshwater areas	-0.07906	0.006352	-12.446	< 0.001
Urban areas	0.091751	0.005148	17.823	< 0.001
Ayr catchment	0.104696	0.00472	22.183	< 0.001
Spey catchment	0.030372	0.00424	7.162	< 0.001
Catchment Scientists	0.350055	0.001388	252.188	< 0.001
Environmental Regulator Staff	0.335782	0.001497	224.377	< 0.001
Farm Advisors	0.413825	0.001446	286.089	< 0.001
Water Regulator Staff	0.375958	0.002527	148.752	< 0.001

4.3.6. Future changes to conflict

A majority (27) of stakeholders stated that, by 2030, there would be changes to the major drivers for conflict between land use, ecosystem service provision and stakeholders in their catchment. When listing likely changes, seven stakeholders identified negative changes due to climate change, such as increased flooding, drought and diffuse pollution as well as threats to biodiversity, particularly Atlantic salmon and alpine bird species such as dotterel and ptarmigan. Other negative likely changes mentioned were increased competition due to the Scottish Government's forestry planting targets and increased financial pressures on farmers and local residents. Eleven participants also mentioned likely positive changes, such as improvements to agricultural and forestry practices (through increased agricultural outputs, reduced pollution from farmland and sustainable forestry), improved payments for ecosystem services, restoration of rivers, woodlands and mining sites, more holistic catchment management and increased communication and cooperation among stakeholder groups.

Ten stakeholders perceived the likely impact of the UK leaving the European Union (Brexit) to be negative in terms of land use, ecosystem service provisioning and stakeholder conflict in their catchment. A commonly held concern was that new UK Government environmental legislation would be less stringent than EU legislation, although eleven stakeholders thought that most environmental legislation would be carried over after Brexit. Five stakeholders stated that they particularly hoped that the EU Water Framework Directive and Bathing Water Directive would not become less stringent after Brexit. Seven stakeholders believed that at a minimum, environmental protection from EU farm payments would be matched after Brexit; however, six others voiced concerns that this was unlikely and would force farmers to make 'hard choices' and make it particularly difficult for those farmers who were on the breadline. Eight stakeholders viewed Brexit as an opportunity for land and water managers to have greater involvement in decision-making and saw opportunities for more targeted farm payments that improve food security and broader ecosystem service provisioning than the current Common Agricultural Policy payments.

4.3.7. Conflict mitigation

Four stakeholders commented that stakeholders already generally work well together, with two stakeholders mentioning their catchment management plan as being pivotal in that respect. When asked to suggest how conflict may be further mitigated, they most commonly (6) replied that improved stakeholder communication and cooperation would help reduce conflict in their catchments. This was followed by improving farm payments (4), such as making them less difficult to apply for and increasing effective management of water margins, and by catchment restoration (4) i.e. improving ecological connectivity and installing natural flood management across the catchments. Other ways to alleviate conflict put forward by multiple stakeholders were research into ecosystem service provisioning and conflict in catchments (2), increasing climate change resilience (2), identifying win-win solutions to issues in the catchments (2), and engaging the local community to become more involved in and benefit from how the catchment is managed (2).

4.4. Discussion

Our proposed methodology provides an innovative and engaging tool for unpacking stakeholder perceptions of conflict and land use competition at the catchment scale. This novel technique captured spatial data through participatory mapping across multiple catchments with varying land use and environmental characteristics and from stakeholders from four key land and water management stakeholder typologies. The heat maps of perceived conflict illustrated the complex combination of local and landscape-scale issues present in the catchments. This enabled quantification of how perceived conflicts differed among different catchments and groups of stakeholders and how modelling of land cover types could predict conflict in the study catchments. What makes this methodology novel is the focus at the catchment level, rather than focusing on socio-political borders, and due to its consideration of a broad range of land and water management issues, rather than focusing on a specific type of conflict. While the methodology was effective as a tool to quantify conflict in the catchments, it also provided broader insights into stakeholder perception which offered several advantages, such as aiding the understanding of views of different stakeholder typologies and how they may oppose each other, as well as identifying wider issues, and allowing examination of the potential drivers of land use conflicts, as well as likely future changes and solutions to the issues.

The mixed method approach allowed for robust results complemented with a rich insight into the complex issues in the study catchments. This underlines the benefits of adding a short survey to help inform interpretation of the mapping outputs (Haworth *et al.* 2016). The combined methodology provided opportunity for validation as well as clarification and elaboration of the mapping results, which are some of the main benefits to using mixed methods in participatory mapping (Brown *et al.*

2017). Another way of validating results from the heatmaps was to count the number of conflicts and drivers mentioned in the questionnaire. In our three study catchments those counts were not statistically different from each other; however, a study with larger stakeholder cohorts may produce statistically significant differences between catchments for that measurement, aiding validation of the heat maps further. The methodology was effective at identifying and characterising spatial hotspots of perceived conflict in catchments of varying land use, environmental characteristics and stakeholder cohorts, which demonstrates the potential transferability of this approach to the rest of the UK, and beyond, as well as to broader groups of stakeholders, such as non-governmental organisations (NGOs) and the public.

Consulting with and engaging the public allows for the elicitation of local knowledge, which can further inform land and water management decision-making while empowering communities and developing social capital (Brown *et al*. 2020). Mapping approaches, such as the one presented here, can also help to bridge the communication gap between lay and expert knowledge (Zolkafli *et al.* 2017). Our methodology could be used to capture views and perspectives of the general public following minor modification, for example by removing or explaining terminology such as "ecosystem services", and by adapting the maps shown to stakeholders to include cities, towns and landmarks, to make it easier to pinpoint areas they may want to highlight. When working with our catchment experts, we assumed that they had a well-established knowledge of places in the catchment. Although this was overwhelmingly the case, stakeholders struggled to immediately locate areas at times, and so the layman may find the catchment maps disorienting without place names. When working with expert stakeholders we would suggest to continue avoiding place names to not draw attention to, and cause bias towards, conflict surrounding urban areas.

4.4.1. Typologies of stakeholders and conflict

Although each of the catchments had distinct local issues and sources of conflict raised by stakeholders, several themes were raised by all four stakeholder groups and in all study catchments. Most of the broader themes emerging from the mapping exercise (forestry, energy provisioning, agricultural diffuse pollution, urban development and flooding) were also identified by all of the four stakeholder groups, whereas pressures on farmland management were only identified by Farm Advisors and Catchment Scientists, and pressures from tourism and recreation were only identified by Environmental Regulator Staff and Catchment Scientists. Hence, Catchment Scientists were the only stakeholder group to identify the entirety of the categories of conflict, which may be a reflection of their role as unbiased outside observers of land and water management issues (Rose & Parsons 2015).

Different typologies of stakeholders did not, however, overlap more within their groups than across all the responses, which was also reflected by congruent reporting of types of conflict by the stakeholder groups. This may indicate that membership to a particular stakeholder typology did not impact on stakeholders' ability to identify a range of land and water management conflicts, and not just those within their main domain of interest. It is still vital, however, to include different actors in any such participatory engagement exercise, as they will have different perspectives on issues and likely solutions to those (Micha *et al.* 2018), despite this not being statistically quantifiable in the overlap analysis. The significantly lower overlap between Water Regulator Staff was likely due to them selecting a smaller number of more specific and local issues, such as flooding or abstraction, which are more relevant in their day-to-day work, as opposed to wider, landscape scale conflicts in which the other three stakeholder groups may be much more engaged in. The four main drivers of conflict were also identified by all stakeholder groups and in all catchments which indicates that, although we were only able to incorporate the views from a limited number of stakeholders, we were able to effectively elicit the general views of the stakeholders in the three catchments.

4.4.2. Land cover data

Results from the regression modelling linking perceived conflict to coastal, grassland and urban land use correspond well to what has been found in other participatory conflict mapping studies: Coastal areas are used by a number of different stakeholder groups and are under multiple and cumulative environmental and socio-cultural pressures (Noble *et al.* 2019; Stepanova 2015). In this study the main underlying issues surrounding coastal areas were bathing water quality, flooding and urbanisation. Urban development may cause rural-urban conflict when resources such as water or valuable farmland are partitioned away from rural areas to secure the needs of a growing urban population (Duvernoy *et al.* 2018; Punjabi & Johnson 2019). Although Scotland is not water scarce, several stakeholders were concerned about aquatic ecological quality being impacted by urban development that already had significant abstractions for hydro power, agriculture and local industries. A common trade-off of renewable hydroenergy is its impact on the ecological needs of the river (Couto & Olden 2018; Lees *et al.* 2016). The spatially constrained nature of urban and coastal areas also likely contribute to the fact that we found conflict in those areas to be highly acute.

Agricultural diffuse pollution has overtaken urban and industrial contamination as the major source of pollutants to water bodies in a number of high-income countries (Evans *et al.* 2019). It is a key contributor to impacts on ecological status and bathing water quality in Scotland, which is likely why improved grasslands were shown to be more likely areas of conflict for the stakeholders of this study (Aitken 2003). Arable land was not seen to be as significant, which may be due to the large impact intensive dairy pastures are having on water quality in the Ayr catchment, whereas in other river catchments arable farming may be considered more at risk of contributing pollutants, and hence more likely to cause conflict.

4.4.3. Temporal aspects of conflict

The major hotspot of conflict identified in the Esk catchment (surrounding the flood mitigation scheme which caused extensive flooding and loss of agricultural soil downstream) demonstrates how a relatively localised issue can be highly relevant for stakeholders across the catchment. It also illustrates how a short and sudden event can have broader temporal impacts on conflict among stakeholders (Saad-Sulonen *et al.* 2018), and shows how an extreme flood event which happened in recent memory can be very significant to stakeholders, which in turn may help develop resilience to future flooding in the catchment (McEwen *et al.* 2017).

As well as capturing a snapshot of current conflict, the methodology was also able to capture some of the emerging and prospective temporal variability by asking stakeholders to consider current issues and likely changes after Brexit and by 2030. Stakeholders identified a number of likely future opportunities and risks for their catchments, for example how a possible departure from the Common Agricultural Policy incentive scheme may allow for more targeted payments for ecosystem services that aim for greatest socio-ecological benefits (Burton *et al.* 2018).

To elicit more of the temporal variability, the mapping exercise could be repeated again in 2-5 years to investigate the realised longer-term outcomes and assess which conflicts were more dynamic. Alternatively, the methodology could be adapted as part of a participatory scenario planning exercise to help foster common understanding and engage stakeholders with future planning of social-ecological systems (Oteros-Rozas *et al.* 2015). Another finding which illustrates the temporal dimension of land use conflict is the historical mismanagement of mines by bankrupted mining companies in one of the study catchments. It highlights how failure in land management can have severe negative impacts decades into the future and illustrates the importance of managing land and water resources responsibly. Other legacy drivers of conflict may also arise not merely due to economic forces, but due to social failures, such as the inability of two stakeholder groups to communicate and build a shared understanding (Paveglio *et al.* 2015). Previously we have shown participants from the Farm Advisor and Environmental Regulator groups had the greatest lack of understanding of the opposing group's interests, which likely exacerbates tensions between the two groups (Stosch *et al.* 2019). However, the increased communication and cooperation between those two stakeholder groups in Ayr, which is a 'diffuse pollution priority catchment', seems to have alleviated conflict, rather than exacerbating it, likely due to an increased rapport and shared understanding among individuals of the two groups as stated by stakeholders in this study.

4.4.4. Drivers of conflict

Legislation was the most commonly mentioned driver behind conflict in all of the catchments and by all participants, which highlights that although supra-catchment policies such as the EU Water Framework Directive encourage bottom-up management and participatory approaches, there is a need to adapt national and EU-wide policy and governance to allow more flexibility and self-determination at the local level to incorporate multiple knowledges and perspectives (Rollason *et al.* 2018). The Scottish Government target of expanding forestry in Scotland by 10 000 ha per year was highlighted as a policy that is likely to exacerbate conflicts and land use competition in all the study catchments, although it has been implemented with a goal of improving overall ecosystem service benefits. When investigating stakeholder views on woodland expansion in Scotland, Burton et al. (2018) similarly found that stakeholders voiced reservations about possible trade-offs from tree planting, although they generally thought it to be an overall positive initiative.

Increases in urban development and tourism was also considered a major driver of conflict, particularly in the Spey catchment. A hotspot was identified by stakeholders in the uplands near the local tourism town, where increases in tourism and recreation are impacting on those "wild" spaces which originally draw people there (Fedreheim & Blanco 2017); demonstrating that tourism development has the potential to be a key driver of conflict (Moore *et al.* 2017). Other studies have also found that an increase in the recreational use of an area can cause conflict with biodiversity conservation (Coppes *et al.* 2017; Karimi & Brown 2017), or between different recreational users (Wilkes-Allemann *et al.* 2015). As such areas are seen to be too often frequented, more people will likely disperse further into the uplands, which may exacerbate impacts on biodiversity in a wider area of the catchment. As people's attitudes towards tourism development can diverge between different segments of a population (Lechner *et al.* 2020), the Cairngorms National Park Authority may want to carry out further research into the nuanced perceptions of space specific conflict in the uplands to inform future national park management to reduce conflict driven by growing tourism and recreation.

4.4.5. Using participatory conflict mapping towards more integrated catchment management

Our methodology has revealed several hot spots of perceived conflict in the study catchments, suggesting that these areas require attention in future policy and management proposals. When looking at relatively explicit and localised conflict, expert judgements informed by our methodology may be sufficient in helping to find solutions to land and water management issues, for example, if adapted to be used as part of a group exercise (Jacobs *et al.* 2015). As many conflicts in catchments are inherently complex, however, the main benefit of this methodology may be as a tool to quantify and spatially identify areas where more focused problem-solving efforts may be needed. This could be achieved through the use of frameworks for managing conflicts, such as was developed for conservation agencies by Young et al. (2016), or ecosystem service-based tools to critically evaluate trade-offs and create a range of solutions to sustainably manage landscapes while considering multiple socio-ecological benefits (Quilliam *et al.* 2015). A combination of the participatory approach we have used here and ecosystem service mapping could also address the lack of sufficient primary, scale-appropriate data and validation studies using modelled data often have (Martínez-Harms & Balvanera 2012; Seppelt *et al.* 2011). Our methodology could also be used on a national scale to identify 'complex conflict priority catchments', similarly to identifying 'diffuse pollution priority catchments', where catchments that are experiencing multiple and exacerbating conflicts, such as the Ayr, could benefit from a more holistic management of landscapes. Such a catchment could then be supported through government funds to help set up ecosystem service-based catchment management initiatives and encourage communication among stakeholders in the catchment. Many of the participants stated that often conflicts arise due to there being a clear trade-off between two reasonable interests which are conflicting, hence making a ‘win-win’ solution unlikely. However, involvement of relevant stakeholders increases the likelihood of measures being implemented and the knowledge of local conditions can aid identification of efficient solutions at lower cost (Graversgaard *et al.* 2017), and stakeholder acceptance appears more related to processes than to outputs, even when final decisions do not reflect all participants' stakes (Kochskämper *et al.* 2016). Increasing dialogue among stakeholder groups would help different parties to understand each other’s view points and lead to stakeholder empowerment and network building aiding future cooperation (Brunet *et al.* 2018; Kochskämper *et al.* 2016).

Our methodology may also be carried out with broader groups of stakeholders, such as NGOs or the public, to form an engaging public participation tool for informing river basin management consultations. As the third and final cycle of the EU Water Framework Directive’s (WFD) River Basin Management Plans end there may be a need to redesign and improve future consultation frameworks. Although participatory processes have been more commonly integrated into the WFD, this is still lacking at appropriate sub-catchment scales where land and water management conflicts arise (Pellegrini et al. 2019). A quantitative review of participation under the WFD concluded that there is a need for determining which engagement instruments are working as current approaches lack evidence of improving good ecological status of catchments (Rimmert *et al.* 2020). Rimmert *et al.* also found that interactive communication in the form of deliberation or dialogue was uncommon and structured methods of knowledge elicitation or aggregation were utilised the least, which shows an opportunity for integrating engaging participatory methodologies such as the one presented here into future river basin management consultations.

4.5. Conclusions

Increasing demands on catchments to supply food, deliver water and energy security and provide wider ecosystem services requires a holistic and sustainable approach to catchment management, which can be challenging. Participatory approaches such as the methodology reported here are crucial to aid the understanding and management of socio-ecological systems and help to reduce conflict among land and water management stakeholder groups. Engaging key stakeholder groups in our study catchments allowed quantification of local and landscape-scale issues as well as insight into stakeholders' perceptions on the current land and water management issues in their catchments of interest. There is an opportunity for innovative and engaging participatory approaches to play a key role in assessing conflict and land use competition and identifying catchments and areas with a particular need for more holistic land and water management, as well as playing a vital part in initiating discourse among stakeholder groups to foster mutual understanding and decision-making.

5. Characterising stakeholder networks in catchments of contrasting water management issues

5.1. Introduction

Global demographic, economic and climatological changes are increasing demand for natural resources, which in turn impacts on ecosystems' ability to function sustainably and remain resilient to shocks and disasters (Hogeboom *et al.* 2021). Society is faced with the complex task of balancing the often opposing demands for water, food and energy security of a growing global population while at the same time protecting biodiversity and mitigating impacts from climate change (van den Heuvel *et al.* 2020). Aquatic, riparian and coastal ecosystems host some of the most diverse biodiversity and supply critical ecosystem services for human health and well-being (Reid *et al.* 2019), but they are increasingly threatened by habitat alteration, water pollution, overfishing, exotic species introduction, fragmentation and flow regulation (Albert *et al.* 2021). Water resources are also highly vulnerable to climate change and by 2050 around 50% of the world's population could be living in countries experiencing water stress (Schlosser *et al.* 2014). Due to hydrological connectivity, river catchments often integrate multiple pressures (Elosegi & Sabater 2013), which makes them less resilient to change and reinforces the concept of the catchment being the most appropriate scale for holistic land and water management (Johnson *et al.* 2017). Catchments integrate land, water and people with diverse roles and views and function as socio-ecological systems; thus, effective catchment management must recognise the importance of stakeholder networks and their influence and interactions in order to ensure long term sustainability that benefits people and catchments.

Managing the breadth of water uses and users within a catchment is a complex task, which often necessitates a role for social, political, and economic institutions of a country (Berger *et al.* 2007) and is further complicated for trans-boundary river basins. Thus, fragmentation of stakeholder networks can arise from shifts in local jurisdiction and in turn lead to less integrated decision-making despite high levels of awareness of shared water management issues across the catchment

(Navarro-Navarro et al. 2017). Different stakeholder typologies (e.g. regulators, water industry practitioners, landowners etc.) within a catchment may also express varying preferences on water management decision-making, depending on their knowledge, values and connections to the landscape (García-Nieto *et al.* 2015). There is hence a growing call for more stakeholder-focused approaches to water resource management to balance the varied and sometimes opposing demands on water resources towards more adaptive and integrated decision-making (Mason et al. 2018; Mohamad Ibrahim et al. 2019).

Stakeholder analysis seeks to identify stakeholders (individuals, groups, or organisations, who can affect or be affected by decision-making in a system), analyse differences among stakeholders, such as their involvement in the decision-making process, as well as investigate relationships among stakeholders (Reed 2008). Stakeholder analysis is also used to understand the diverse range of potentially conflicting stakeholder interests (Prell *et al.* 2007; Stosch *et al.* 2019). In social network analysis, actors in a social network are depicted as nodes and links are established to other actors, allowing the analysis of the relations between nodes to identify the most influential actors and in contrast those at the periphery of the network. Both stakeholder analysis and social network analysis have been demonstrated as useful tools in natural resource management (Ahmadi *et al.* 2019). In terms of water management, stakeholder and social network analysis have been used to analyse the structure of water governance networks (Horning *et al.* 2016), identify their spatial scale mismatches (Sayles & Baggio 2017) and highlight opportunities for cooperation within them (Luzi *et al.* 2008), or to analyse catchment stakeholders' interests and spheres of influence (Ogada *et al.* 2017). Stakeholder network analysis has also been used to find ways to improve fishery commission management (Mulvaney *et al.* 2015), mitigate impacts of climate change on water management (Yang *et al.* 2018), and identify the social stability risk of large hydro engineering projects (He *et al.* 2018). One of the main drawbacks of common stakeholder mapping techniques is that they tend to identify the 'usual suspects' and there is a danger that this may lead to the under-representation of peripheral stakeholders (Reed *et al.* 2009). Engaging stakeholders involved in water management for social network analysis and stakeholder analysis offers potential to facilitate effective knowledge exchange among stakeholders in a network, capitalising on important expertise and understanding as well as highlighting differences in how stakeholders perceive and value other groups (Darvill & Lindo 2016). This presents a gap in the research where novel techniques may be developed for ameliorating conflicts, fairly representing diverse interests and preventing further marginalisation of under-represented groups.

5.1.1. Governance structure of Scottish catchment management stakeholders

A number of different stakeholder sectors impact on catchment management decision-making in Scotland (Fig. 5.1). In terms of the governing sector, the European Commission and the Scottish Government issue the most legislation relating to land and water management, such as the EU Water Framework Directive, the EU Bathing Water Directive and the Scottish Government's Climate Change Plan aiming to increase forest planting and peat restoration. Scottish Water (Scotland's public water and wastewater company) and the Scottish Environment Protection Agency (SEPA) represent key regulatory stakeholders due to their central role in enforcing land and water management in Scotland. Of the private businesses that impact catchment management, farmers play a crucial role as they are a major abstractor of water for irrigation, but mainly due to loss of farm nutrients and chemicals continuing to be the major cause of water courses failing to meet EU environmental standards in Scotland and the rest of Europe (European Environmental Agency 2018). Timber production can also intercept and acidify water and contribute to diffuse pollution with sediments, fertilisers and pesticides (Sing *et al.* 2018). Energy companies strongly influence the water environment through hydropower scheme management, mining activities and wind farm development (Evans *et al.* 2010). The food and drinks industry, which has the highest rate of water use in the UK industrial sector (Ajiero & Campbell 2018), is a major contributor to Scotland's economy, contributing around £14 billion annually and accounting for one in five manufacturing jobs. This industry impacts on catchments through their use of potable water and due to impacts from salmon farming and other fisheries management. The tourism industry contributes around £6 billion to Scottish GDP, currently representing about 5% of total Scottish GDP. Conservation NGOs, and especially the RSPB have an influence on decision-making through large land ownership in Scotland and lobbying (McMorran & Glass 2013). Education and research is relevant in expanding our knowledge of catchment systems and educating people about how best to manage them. River and Fisheries Trusts are also a significant interest group in Scotland as fisheries are an important national resource and as they also support education and implement mitigation measures (Malcolm 2011).

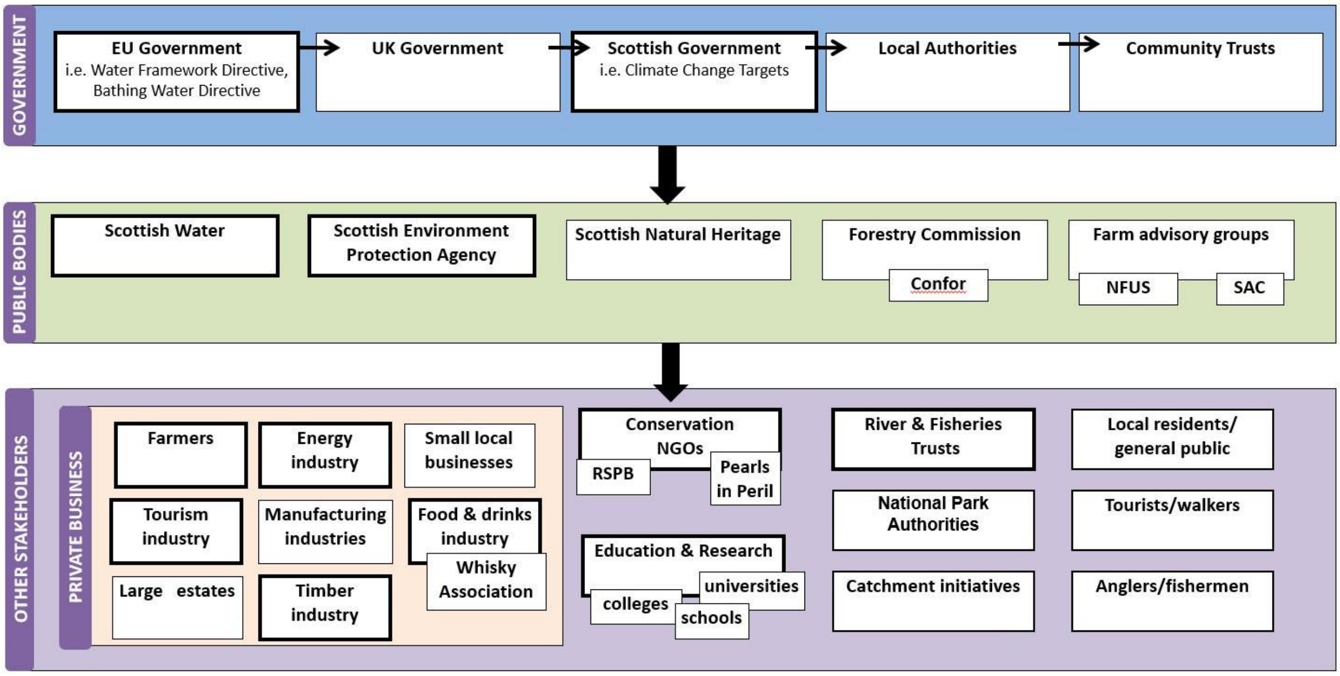


Fig. 5.1: Flow chart of key stakeholders that have an impact on catchment management in Scotland. Boxes outlined in bold indicate stakeholders which likely have a larger influence on catchment management.

Here we combine stakeholder analysis and social network methodologies to elicit perceptions from four key stakeholder typologies involved in water management across three diverse study catchments. Using this approach, the overarching aim of this study was to characterise how influential different stakeholders were perceived to be with respect to catchment management. We used the data from participants' self-reported social networks to provide further insight into the co-occurrence of stakeholders (i.e. how often groups were named together by one participant), allowing core and periphery analysis of which groups had the greatest co-occurrence. This approach represents a novel and rapid methodology for characterising stakeholder networks at the catchment scale and could be used as a complementary methodology for enhancing other stakeholder mapping techniques that are commonly topic-driven rather than catchment-centric.

Each actor (or node) in a network can perceive a network structure subjectively, or as a cognitive social structure (Krackhardt 1987). Such a network can also be constructed objectively with empirical data to compare with subjective perceptions of the network, however, cognitive social structures are data in their own right, giving insights into insider actors' views.

Therefore, the objectives were to: (1) assess whether participants representing four key stakeholder groups perceive the importance of stakeholder influence in line with the existing governance structure of Scottish catchment management; (2) determine which stakeholders are more central to a catchment management network and which are perceived as peripheral; (3) quantify which stakeholders are perceived to have the largest impact as well as the most positive or negative influence on the water environment; and (4) compare outputs between the four participant groups and the three contrasting river catchments.

5.2. Methodology

5.2.1. *Study catchments*

Three catchments from across Scotland were selected on account of their diverse and contrasting geomorphologies, land cover types, stakeholder communities, and land and water management pressures (Figure 2). The River Spey in the north-east, the South Esk in the east and the River Ayr catchment in the south-west of Scotland. The catchments vary in size from ~600 km^2 (South Esk and Ayr) to just under 3000 km^2 (Spey). The River Spey and South Esk catchments are dominated by moors and heathland, followed by sparsely vegetated land in the mountainous areas of the Spey (23%) and arable land in the Esk catchment (31%). Dairy production is a key local industry in the Ayr catchment with pasture accounting for 39% of the land cover. In general, the uplands of the three catchments are dominated by rough grazing, commercial forestry, and sporting estates, while the lowlands accommodate arable land and improved grazing. Tourism and angling represent important local industries, with whisky production also being significant, particularly in the Spey. There are competing pressures on water resources in all three catchments via diffuse pollution from farming practices and point source inputs from sewage discharge, in addition to abstraction for potable water, large hydropower schemes, food and drink manufacture and irrigation.

To simplify, the Ayr catchment may primarily be characterised as an agricultural (and particularly a dairy) catchment, while the Spey could be characterised as a recreation-focused catchment partly within a national park, and the Esk as a mixture of farming (particularly arable), forestry and recreational land uses.

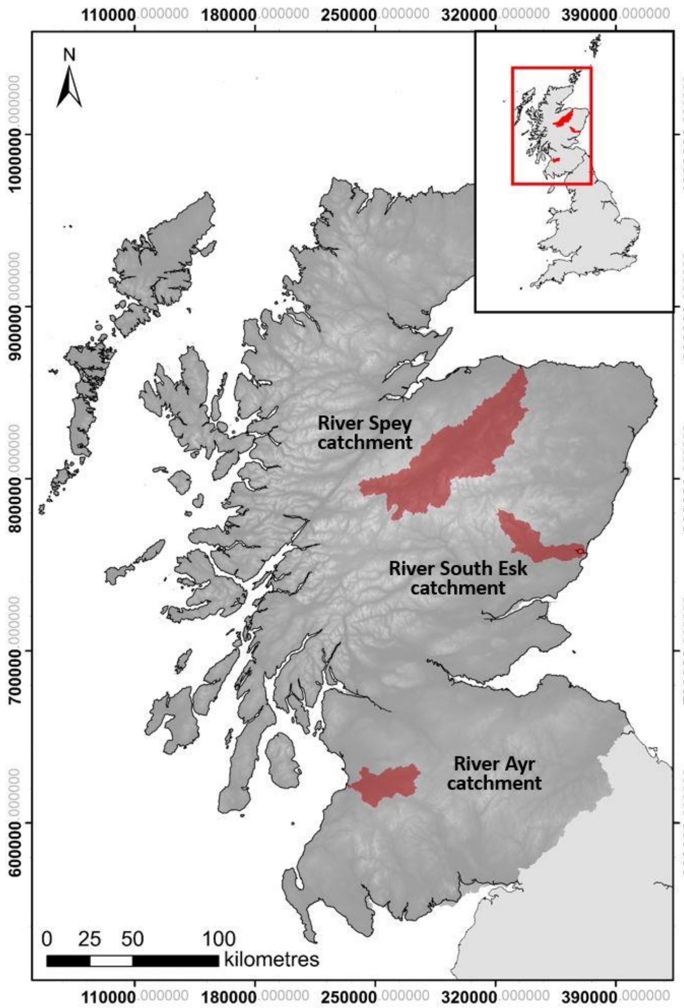


Figure 5.2: Location of the three study catchments in Scotland, UK.

5.2.2. *Sample selection and design of engagement exercise*

In each of the three study catchments, three to five individuals were recruited from four key stakeholder groups: environmental regulator staff (n=12), water industry staff (n=9), catchment scientists (n=11) and farming representatives (n=11). A total of 43 stakeholders carried out the engagement exercise in 2017 and contributed their local knowledge on catchments within which they worked: 15 contributed to the River Spey catchment analysis, 13 to the South Esk and 15 to the Ayr catchment. The full details of the stakeholder sample selection and interview approach can be found in (Stosch *et al.* 2019).

The engagement exercise presented here was part of a larger conceptual modelling exercise in which participants were asked to rank ecosystem services in their catchments, identify various pressures on their river catchment as well as name which remediation measures were already in place. After that, participants were asked to list all stakeholder groups (herein referred to as stakeholders) that have an influence on catchment management within their catchment of interest. Participants were not given a list of possible stakeholder typologies but were asked to recall stakeholders from memory, which helped to inform analysis of which stakeholders were omitted. After completing their list, participants were asked to then state whether they believe each stakeholder to have either a small, medium or large influence on the management of the catchment and whether that perceived influence is positive, negative or overall neutral.

5.2.3. Data analysis

The results from the survey were collated into two matrices; one matrix that captured each participant's perceived size of each stakeholder's influence (either a 1, 2 or 3 for small, medium or large influence, respectively), and another matrix depicting the perceived value of each stakeholder's influence (either a 1, 2 or 3 for negative, neutral or positive influence, respectively). As participants were not given an a priori list of stakeholders to select from, the large number of elicited stakeholders (71) were collated, creating a total of 28 typologies. This was to increase legibility of the stakeholder networks and to group together similar stakeholders, such as NGOs, small local businesses and diverse industry. Any stakeholder group that was named by only a single participant was omitted from the analysis. Both matrices (perceived size of the influence and perceived value) were imported to UCINET 6 for social network analysis (Freeman *et al.* 2015). As the data was collected as a 2-mode valued network we used 2-mode Centrality to calculate the degree score. A 2-mode Categorical Core/Periphery Model was used to separate stakeholders into a core and periphery and a Conversion Projection method was used to turn the 2-mode data into a 1-mode affiliation matrix. A two-way ANOVA was carried out using SPSS version 28 to compare responses (mean degree scores, mean no. of responses, mean perceived influence and mean proportion of negative ties) among the different stakeholder groups and catchments (IBM 2021).

5.3. Results

The 43 participants identified a combined total of 28 different stakeholder groups. As stakeholders were identified by multiple participants, 490 individual scores were collated. On average each participant named 11 stakeholders and the exercise took around 15 minutes per respondent. The Environmental Regulator and Local Government were named most often (35 and 27 participants, respectively). Although the Devolved Government (15) was also mentioned widely, neither the UK Government nor European Commission were mentioned by the participants more than once and were hence not included in the analysis. When collating all the participant responses for all stakeholders, most of the stakeholders received positive scores (305), followed by neutral (125) and negative (60). Similar numbers of stakeholders were identified as having medium (193) and large (191) influence and less as having a small influence on the catchment (106). The Devolved Government (2.67 ± 0.15), Farmers (2.57 ± 0.13), and the Environment Agency (2.48 ± 0.12) had the greatest mean size of perceived influence (Figure 5.3).

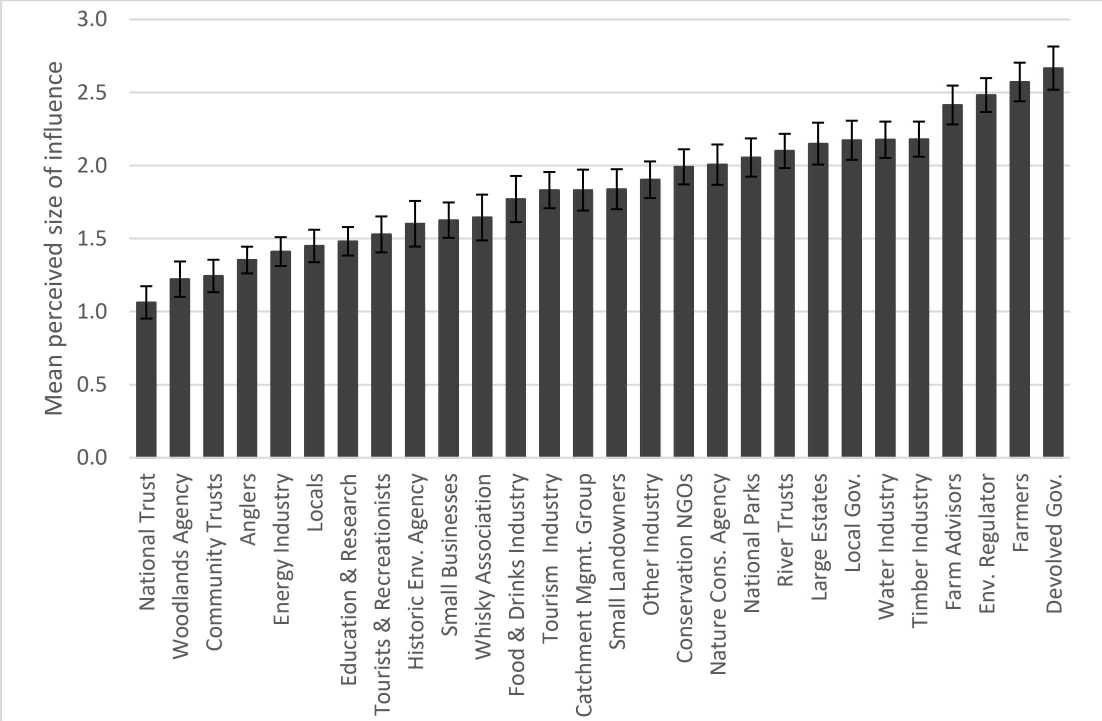


Figure 5.3: Ranked mean size of the perceived influence of the elicited stakeholders on catchment management in the study catchments (± 1 standard error; 1 = small, 2 = medium, 3 = large perceived influence).

The Environmental Regulator was elicited most frequently (35 times), followed by Local Government (27), Water Industry (26) and River Trusts (25, Figure 5.4). Farmers (0.21), the Public Water Agency (0.19), and Timber Industry (0.14) had the greatest proportion of perceived negative influence. When combining responses for negative and neutral influence, Farmers (0.51), Timber Industry (0.44) and Farm Advisors (0.42) scored highest (Table 5.1). Participants belonging to different stakeholder typologies named a comparable number of stakeholders (Table 5.2). Participants responding for the south Esk catchment named the fewest mean numbers of stakeholders (around 10), whereas it was around 12 in the Ayr catchment and around 13 in the Spey catchment.

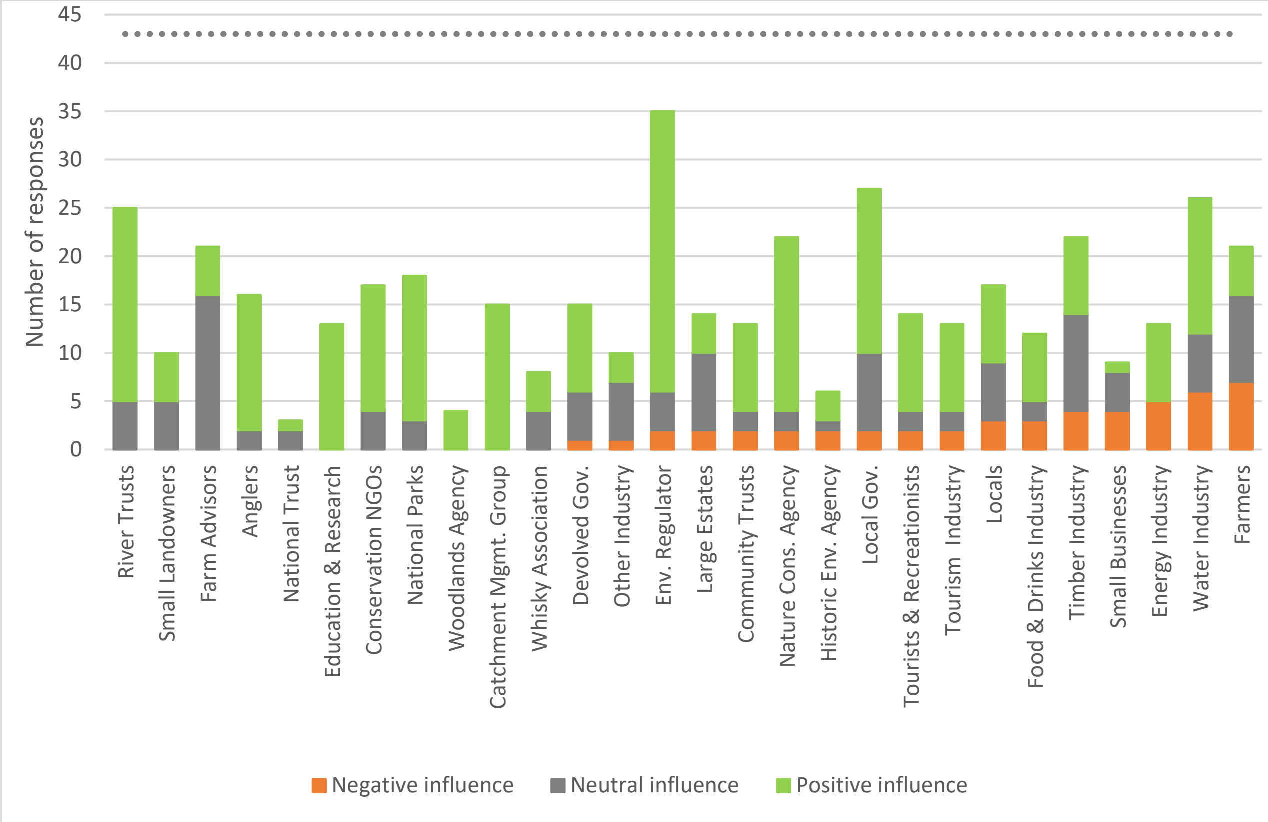


Figure 5.4: Number of responses of perceived negative, neutral or positive influence of the stakeholders named by participants, ranked by perceived negative influence (n=43, dotted line).

Table 5.1: Degree, mean perceived influence (± 1 standard error), and proportion of perceived negative influence and negative and neutral influence of the 28 elicited stakeholders.

	Degree	Mean perceived influence ± st. error	Proportion of perceived negative influence	Prop. of perceived neg. & neutral influence
Farmers	1.81	2.57 ± 0.13	0.21	0.51
Environ. Regulator	2.51	2.48 ± 0.12	0.07	0.16
Water Industry	1.58	2.18 ± 0.12	0.19	0.33
Large Estates	0.81	2.15 ±0.14	0.05	0.26
Timber Industry	1.58	2.18 ± 0.12	0.14	0.44
River Trusts	1.60	2.10 ± 0.12	0	0.12
Community Trusts	0.47	1.24 ± 0.11	0.02	0.05
Nature Cons. Agency	1.19	2.01 ± 0.14	0.05	0.07
Small Businesses	0.47	1.63 ± 0.12	0.12	0.21
Small Landowners	0.51	1.84 ± 0.14	0	0.14
Historic Env. Agency	0.28	1.60 ± 0.16	0	0.02
Farm Advisors	1.53	2.41 ± 0.13	0	0.42
Locals	0.88	1.45 ± 0.11	0.12	0.28
Local Gov.	1.56	2.17 ± 0.13	0.07	0.26
Devolved Gov.	1.19	2.67 ± 0.15	0.05	0.16
Tourists & Recreationists	0.60	1.53 ± 0.12	0.07	0.14
Anglers	0.58	1.35 ± 0.09	0.00	0.07
Energy Industry	0.51	1.41 ± 0.10	0.12	0.14
Other Industry	0.67	1.90 ± 0.13	0.05	0.23
Food & Drinks Industry	0.49	1.77 ± 0.16	0.05	0.07
National Trust	0.07	1.06 ± 0.11	0	0
Education & Research	0.56	1.48 ± 0.10	0	0
Conservation NGOs	0.79	1.99 ±0.12	0	0.05
National Parks	0.84	2.05 ± 0.13	0	0.07
Woodlands Agency	0.12	1.22 ± 0.12	0	0
Tourism Industry	0.63	1.83 ± 0.12	0.05	0.07
Catch. Mgmt. Group	0.70	1.83 ± 0.14	0	0
Whisky Association	0.23	1.64 ± 0.16	0	0.05

Table 5.2: Mean number of responses, mean degree, perceived influence, and proportion of perceived negative influence and negative and neutral influence of the three contrasting catchments and four key water management stakeholders (± 1 standard error). None of these statistics were significantly different between catchments or stakeholder groups (significance of differences between the means at 0.05 level).

	Ayr catchment	South Esk catchment	Spey catchment	Catchment Scientists	Farm Advisors	Environmental Regulator Staff	Water Industry Staff
Mean number of responses ± st. error	11.53 ±0.90	9.92 ±1.34	12.53 ±0.98	11.27 ±1.36	11.18 ±0.88	11.55 ±1.22	11.44 ±1.68
Mean Degree ± st. error	1.02 ±0.10	0.87 ±0.13	1.15 ±0.09	1.03 ±0.13	0.95 ±0.10	1.06 ±0.11	1.03 ±0.18
Mean influence ± st. error	2.45 ±0.09	2.46 ±0.11	2.58 ±0.08	2.58 ±0.09	2.38 ±0.14	2.55 ±0.08	2.48 ±0.10
Proportion of perceived negative influence ± st. error	0.04 ±0.02	0.04 ±0.02	0.02 ±0.01	0.02 ±0.01	0.06 ±0.02	0.02 ±0.01	0.03 ±0.02
Prop. of perceived neg. & neutral influence ± st. error	0.11 ±0.01	0.08 ±0.02	0.11 ±0.02	0.11 ±0.02	0.10 ±0.02	0.10 ±0.03	0.10 ±0.02
Core/Periphery fit (correlation)	0.912	0.889	0.897	0.864	0.861	0.902	0.872

UCINET determined the stakeholders that were most commonly connected in all of the study catchments and hence had the greatest degree score (node size) and those which formed the core of the stakeholder network (Fig. 5.5; pink nodes). The Environmental Regulator had the largest degree score (2.52), followed by Farmers (1.81) and River Trusts (1.60), which were closely followed by Water Industry (1.58), Timber Industry (1.58), Local Government (1.56), and Farm Advisors (1.56). The Devolved Government and Nature Conservation Agency both had a degree score of 1.19. These nine stakeholders were classed within the core and the remaining 19 stakeholders were classed within the periphery (Core/Periphery correlation coefficient = 0.9211). The stakeholders with the highest degree scores within the periphery were Locals (0.88), National Parks (0.84), Large Estates (0.81), Conservation NGOs (0.79), and Catchment Management Groups (0.70). Ties (connections between stakeholder nodes) depict co-occurrence of stakeholders in participant responses. The

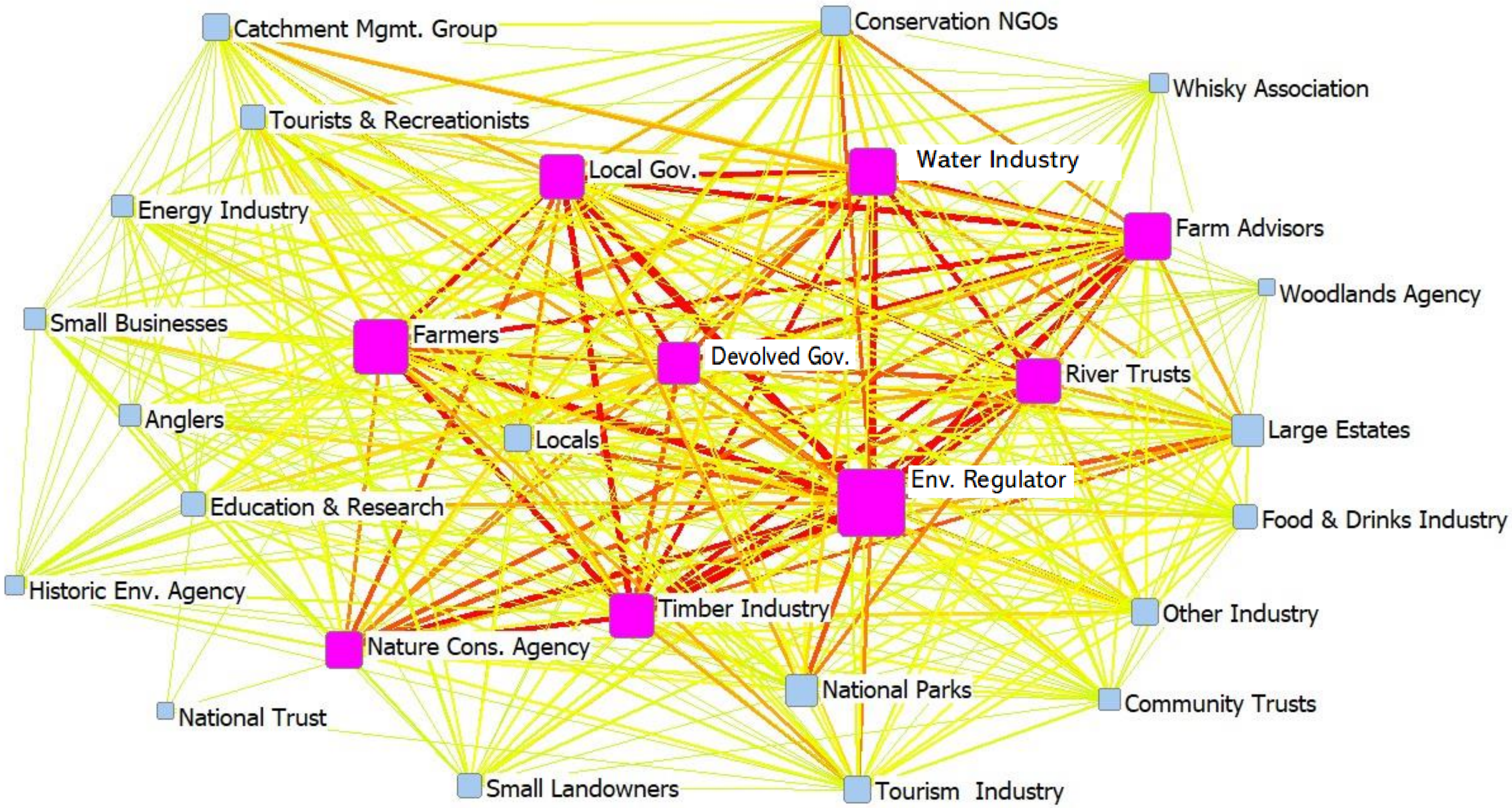


Figure 5.5: One-mode network of the stakeholder typologies, with node size depicting degree score and node colour showing the core (pink) and periphery (blue). Core/Periphery fit (correlation) = 0.9211. Ties depict co-occurrence of stakeholders from participant responses and tie strength is visualised both by line thickness as well as by colour with green depicting low tie strength (max. = 192).

greatest ties are among the core stakeholders; however, there are also moderately high ties between National Parks and the Environmental Regulator and River Trusts, Conservation NGOs and the Environmental Regulator and Farm Advisors, Catchment Management Groups and Local Government and the Water Industry, the Tourism industry and several stakeholders, and Large Estates and the Timber Industry, the Environmental Regulator and several other stakeholders.

The core stakeholders from the Ayr catchment were identical to the Esk's eight nodes, but also included the Nature Conservation Agency (Figure 5.6). Responses from the Spey catchment identified ten core stakeholders. This included seven that were also selected in the Ayr catchment, but not the Devolved Government and Farmer nodes, unlike the core groups in both the Esk and Ayr catchments. The analysis also included three groups in the Spey network's core that were lacking in the other two catchments: Large Estates, Conservation NGOs and National Parks. The Whisky Association was only named in the Spey catchment and National Parks were named in the Spey and Esk, but not in the Ayr catchment. The Ayr was the only catchment where the Environmental Regulator did not have the greatest degree score, with Farmers scoring slightly higher.

When comparing the core/periphery analysis between the four stakeholder typologies that the participants belonged to, networks of the Water Industry Staff had the greatest number of nodes within the core (11), followed by Environmental Regulator Staff (9), Catchment Scientists (8) and Farm advisors (6) (Figure 5.7). Stakeholder nodes that were classed as being within the core across all four participant typologies were the Environmental Regulator, Local Government, River Trusts and the Timber Industry. The Water Industry, Farm Advisors and Farmers were classed as being within the core by three of the four participant groups. The Devolved Government, the Nature Conservation Agency and National Parks were classed within the core by two participant groups and Conservation NGOs, the Food and Drinks Industry and Large Estates by one. Farm Advisors were the only

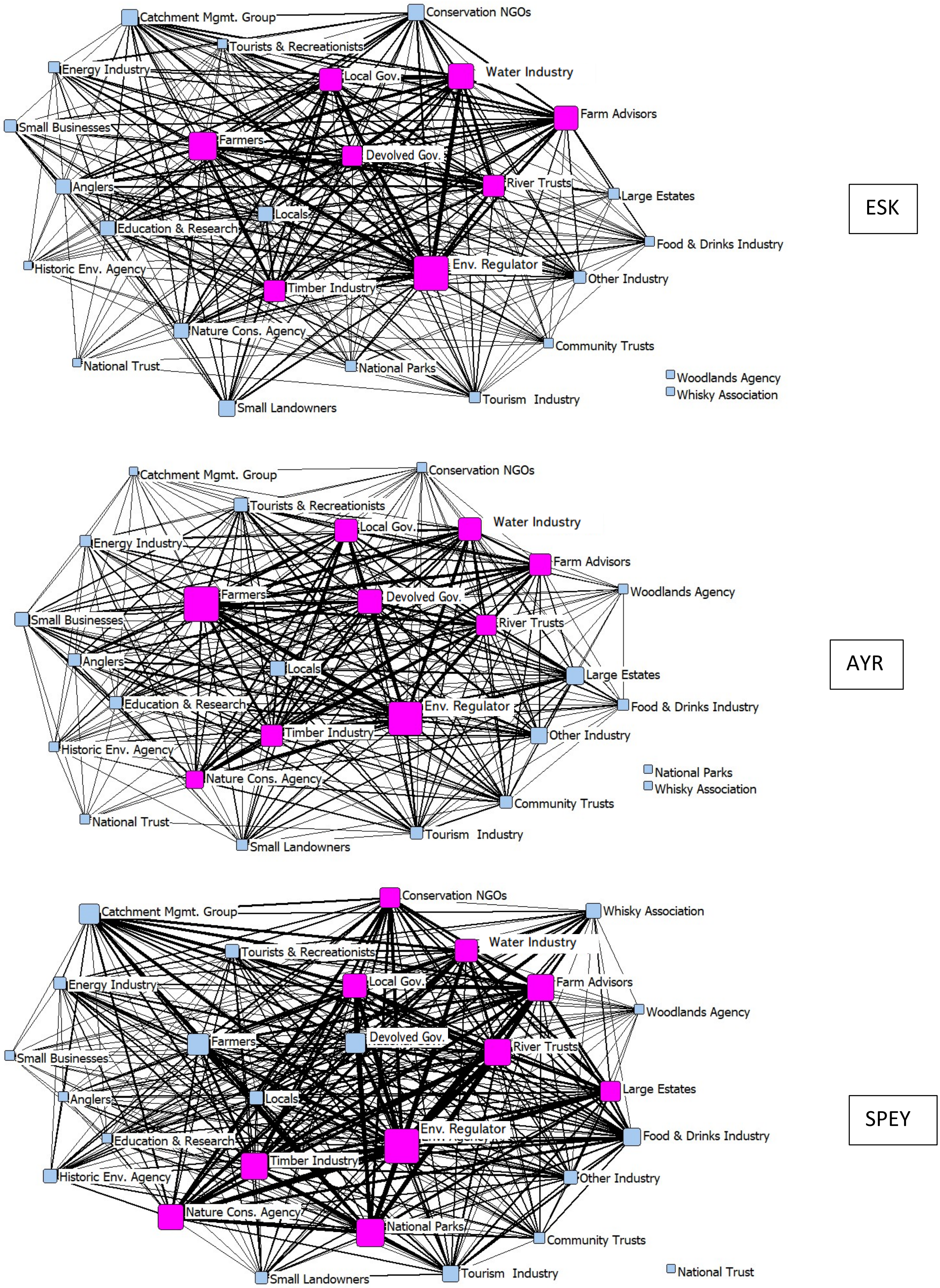


Figure 5.6: The three study catchments' one-mode networks, with node size depicting degree score and node colour showing the core (pink) and periphery (blue). Core/Periphery fit (correlation) = 0.889 (South Esk), 0.912 (Ayr) and 0.897 (Spey).

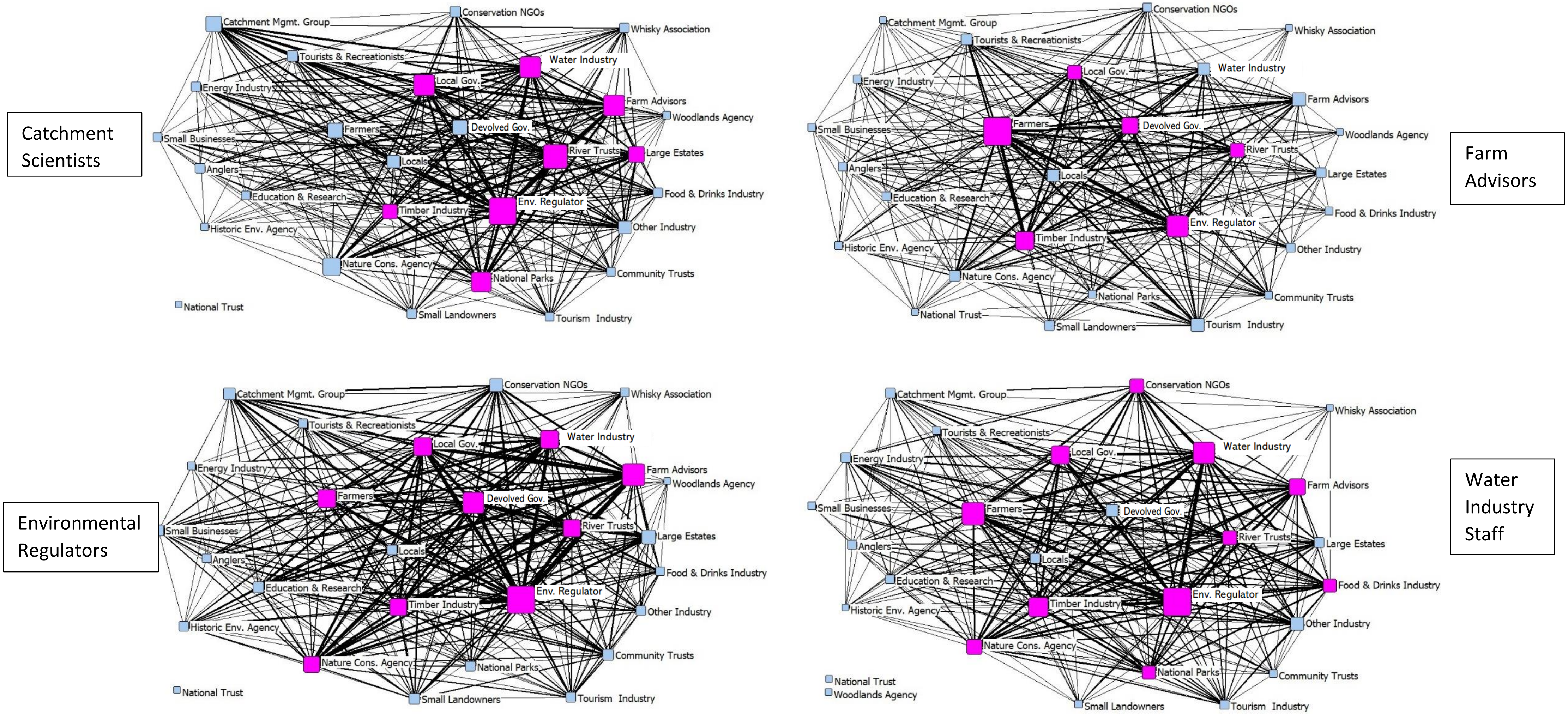


Figure 5.7: The stakeholders' one-mode networks, with node size depicting degree score and node colour showing the core (pink) and periphery (blue). Core/Periphery fit (correlation) = 0.864 (Catchment Scientists), 0.861 (Farm Advisors), 0.902 (Environmental Regulators) and 0.872 (Water Industry Staff).

participant group where a stakeholder (Farmers) had a greater degree score than the Environmental Regulator.

Figures 5.8 and 5.9 show all of the participant's responses, i.e. which stakeholders they listed and whether they perceived that group's influence on their catchment as small, medium or large and whether they thought this influence was overall positive, neutral or negative. Three participants identified National Parks as a relevant stakeholder in the Esk catchment, while twelve did for the Spey and none for the Ayr catchment. Respondents in the Spey catchment chose negative ties half as often as those in the other two catchments and Farm Advisors responded with the greatest numbers of negative ties out of all the participants, two and three times higher than Water Industry Staff and three times greater than Environmental Regulator Staff and Catchment Scientist (Figure 5.9 and Table 5.2).

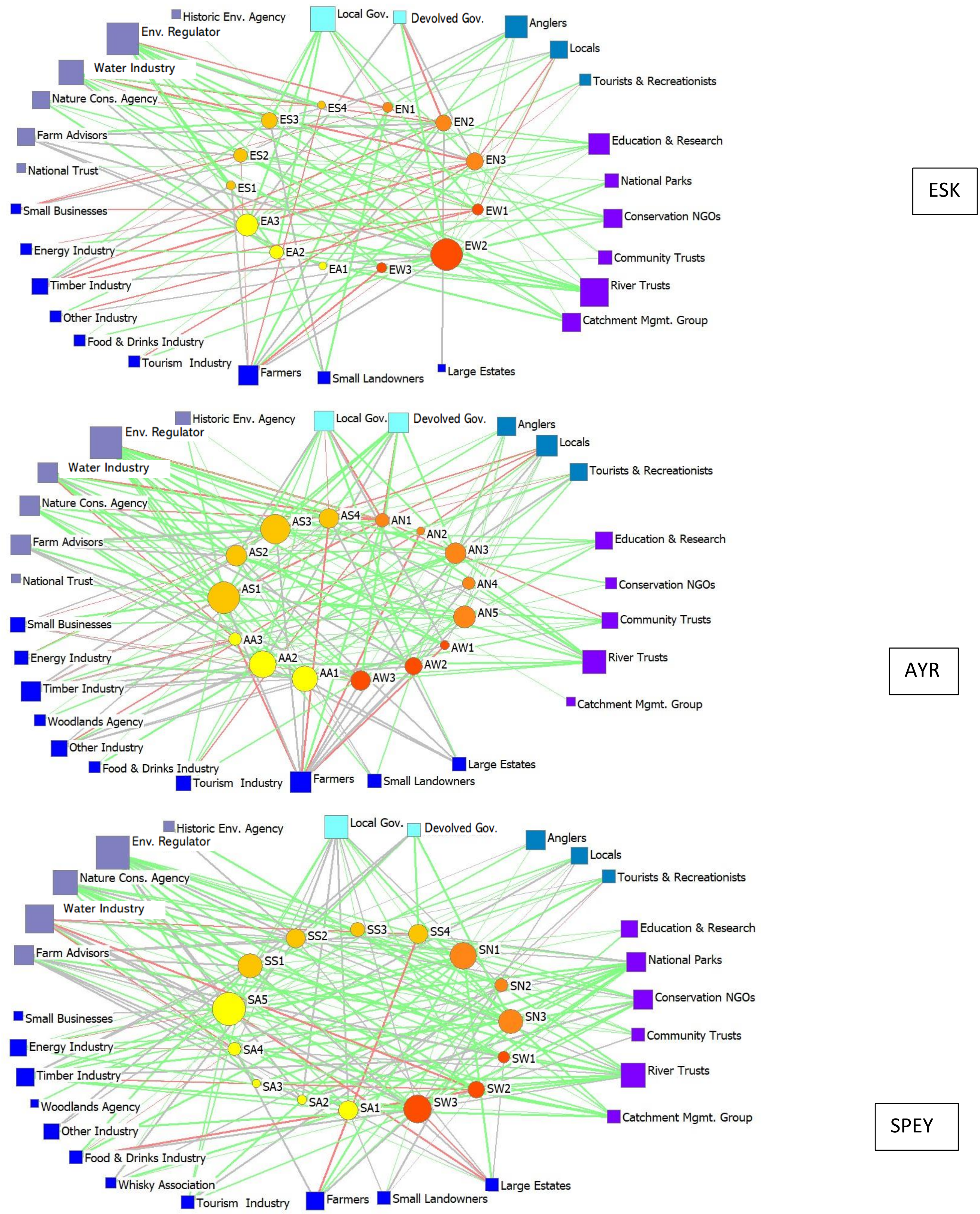


Figure 5.8: The three study catchments' two-mode networks of the stakeholder typologies (squares, coloured to distinguish government (turquoise), public bodies (grey), private business (blue), other stakeholders (purple) and individual actors (cyan)) and participants (circles, yellow (Catchment Scientists), amber (Environmental Regulators), light orange (Farm Advisors) and dark orange (Water Industry staff)) with tie strength depicting a small, medium or large influence on the catchment and tie colour showing a negative (red), neutral (grey) or positive influence (green). Node size depicts degree score.

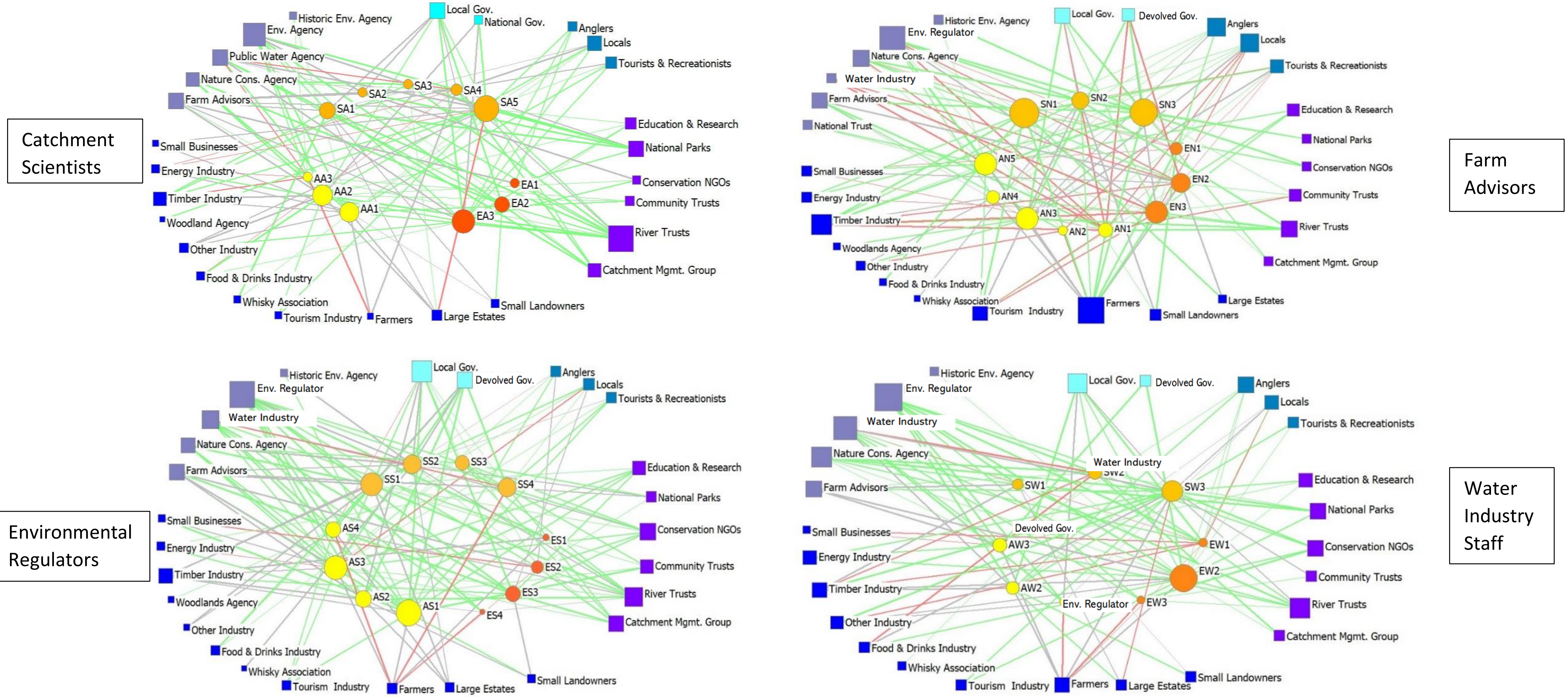


Figure 5.9: The stakeholders' 2-mode network of the stakeholder typologies (squares, coloured to distinguish government (turquoise), public bodies (grey), private business (blue), other stakeholders (purple) and individual actors (cyan)) and participants (circles, yellow (Ayr), amber (Spey) and orange (Esk)), with tie strength depicting a small, medium or large influence on the catchment and tie colour showing a negative (red), neutral (grey) or positive influence (green). Node size depicts degree score.

5.4. Discussion

We combined stakeholder analysis and social network methods to provide a novel stakeholder-mapping tool capable of identifying important distinctions between interactions among the land and water management communities across three contrasting study catchments. The methodology used a participatory approach to identify the perceived importance of key stakeholders and how connected they were, and in turn helped to understand which stakeholders were considered core versus peripheral with respect to catchment management. Stakeholder mapping exercises are often undertaken without direct stakeholder input and therefore may reflect the biases of the researchers with regard to who they believe has most influence and interest concerning a particular subject area rather than the perceptions of the stakeholders themselves, leading to questions about the legitimacy of how stakeholders are mapped and characterised (Reed *et al.* 2009). The catchment-scale stakeholder mapping approach reported here therefore represents a stakeholder-driven framework for identifying key players in catchment management. By reporting on perceived influence of stakeholders across three different catchments accommodating varying management issues we provide a framework that is transferable and offers advantages in that the exercise provides a wealth of data, while it is simple, engaging and quick to carry out for each of the participants.

We engaged stakeholders in three contrasting catchments, one characterised by dairy production, one with mixed arable, rough grazing and forestry land use and a catchment partly within a National Park. The differences between land use among the three catchments corresponded well with how Farmers, the National Park and other stakeholders were represented in the stakeholder maps, indicating that the methodology can differentiate between catchments, making the methodology accurate and transferable. When using this methodology in practice for catchment management, a much smaller number of participants would likely give a good insight into the stakeholder network. Here we wanted to be able to statistically compare responses among the stakeholder typologies in each catchment and hence elicited responses from at least three participants for each group. The networks depicting only results from one stakeholder typology are based on as little as nine responses, but still have high Core/Periphery fit correlation scores. When adapting this methodology, initially, responses from an even smaller number of key stakeholders could be analysed to inform whether more responses are needed (if variance between responses is high), or whether a major core stakeholder has not been invited to participate. This makes the methodology a useful stakeholder mapping exercise, which can complement an existing portfolio of stakeholder mapping tools in stakeholder identification and in investigating their relationships (Raum 2018).

5.4.1. Contrasting catchments and their stakeholder typologies

The number of participants mentioning National Parks as a relevant stakeholder corresponded to the catchments' area located within the Cairngorms National Park. A majority of the area of one catchment is within the park's borders, while another accommodates only a small area of the National Park and the third catchment is located outside of any national parks. The Nature Conservation Agency and Conservation NGOs were also more influential and part of the core in the touristic catchment. This correlates with significant landownership of NGOs in the catchment area and habitat provision for endangered species such as capercaillie, Scottish wildcat and golden eagle. A stakeholder only mentioned by participants in that catchment was the Whisky Association, which corresponds to the large number of Whisky distilleries located along the River Spey. The Environmental Regulator was elicited most often, which reflects the Environmental Regulator's clear role at the heart of central catchment management issues such as diffuse pollution and flooding. The stakeholders with the largest numbers of perceived negative influence were also associated with the two main sources of pollution to watercourses: waste water inputs and diffuse pollution from agriculture. These results suggest that the methodology was able to identify differences in underlying issues in land and water management in the study catchments.

5.4.2. Perceived magnitude and value of stakeholder influence

The Devolved Government, Farmers and the Environmental Regulator were perceived to have the largest influence on catchment management, which was consistent within the three study catchments. This shows, again, the central role of the Environmental Regulator in enforcing land and water management in Scotland. Although much of the legislation protecting water resources comes from the EU, water is a particularly critical resource for all sectors of the Scottish economy, such as for manufacturing, energy, agriculture, food and drink and tourism, and the Scottish Government acknowledges this through their 'Hydro Nation' agenda (Greig & Rathjen 2021). Farmers are also central to catchment management in Scotland due to their use of water as well as due to diffuse pollution from agricultural land which continues to represent a wide-scale and persistent problem in many regions of Scotland (Scottish Government 2016). Farmers also had the greatest proportion of perceived negative influence and the greatest proportion of the sum of neutral and negative influence. This measure was added to the results as several participants voiced that they felt uncomfortable about ascribing the term 'negative' to any stakeholder, which sometimes then led them to select the 'neutral' option. If this survey was to be repeated it may be more effective to ask whether the influence of each stakeholder was positive, neutral or negative concerning a specific measurement, such as 'Ecological Status' which is well defined under the EU Water Framework Directive and key stakeholders would be very familiar with. When comparing negative ties in each of the catchments, participants from the Spey catchment identified half as many negative ties than those in the other two study catchments. This may link back to greater cooperation and understanding among stakeholders in this particular catchment or lower levels of conflict as was identified in our previous conflict hotspot research in the study catchments (see Chapter 3). Negotiation and joint learning helped to foster bridging social capital between farmers and Dutch government officials that increased shared views on conservation goals (Westerink *et al.* 2017). The Spey catchment was also the only study catchment where Farmers were not classed as a core stakeholder. Perhaps this reflects a smaller influence of agricultural land management in the catchment causing less negative impacts on aquatic quality resulting in fewer perceived negative ties. The Spey catchment has a large proportion of sparsely vegetated mountainous areas (23%) and moors and heathland (29%) and smaller proportions of pasture (9%) and arable land (2%) than the other two study catchments.

Farm Advisors were up to three times as likely as other participants to select negative ties and none of the Farm Advisor participants perceived Farmers as having a negative influence on catchment management. Farm Advisors are likely to have differing attitudes on what constitutes a 'good farmer' relative to other catchment stakeholders as they are more understanding of farmers' landscape values while they also assist them to comply with the increasing environmental legislation. As society's expectations of the farming community change from that of supporting food security and animal welfare issues to inclusion of broader social and environmental goals (Saunders 2016), farmers may not be necessarily opposed to specific practice changes to protect environmental quality, but may be resistant to challenges to their identity of what makes a 'good farmer' (Collins 2018). Although there may also be scepticism from farmers regarding diffuse pollution control programmes and their efficacy in Scotland (Barnes *et al.* 2009), there may be a shift in how riparian environments contribute to what constitutes a 'good farmer' (Thomas *et al.* 2019).

Another cause of tension among stakeholders with regard to land and water management may extend to opposing views on how their own and other stakeholders are being perceived (Stosch *et al.* 2019), and be more complex than simply accommodating differing attitudes about land and water management in general. When stakeholders are required to work together negative stereotyping, distrust and scapegoating may arise, causing conflict and threatening the social harmony of collaborative systems (Curșeu & Schruijer 2017). Adapting the approach used here in one-to-one interviews for the context of a group discussion could present an opportunity for stakeholders to articulate their views in a non-confrontational and abstract setting as well as reflect on how accurately the data represents stakeholder networks in their catchments (Cebrián-Piqueras *et al.* 2017). In doing so it could promote discussion of otherwise implicit attitudes, build shared mutual understanding and facilitate future cooperation (Brunet *et al.* 2018). The UK leaving the EU and being able to devise their own agri-environment schemes may be an opportunity to involve stakeholders in their design and to allow farmers to embed their understanding of landscape stewardship and their landscape values (Raymond *et al.* 2016). Other stakeholder engagement exercises, such as stakeholder Delphi analysis and fuzzy cognitive mapping could also benefit from using 'insider knowledge' from key stakeholders, such as is presented here, instead of relying on desktop study to select stakeholder participants (D'Agostino *et al.* 2020).

5.4.3. Comparing perceived importance of stakeholders to hypothesised outcome

Participants recognised the importance of stakeholders similarly to the hypothesised diagram of influential stakeholder typologies in Fig 5.1. Comparing the core nodes from all respondents to the hypotheses in Section 5.1.1., a few stakeholders were identified less often as was predicted, e.g. Food and Drinks Industry and Education and Research and Energy Industry. However, a notable difference in expected and elicited results was that neither the UK nor the EU Commission was mentioned by more than one stakeholder across all three catchments. Identifying which stakeholders are missing from studies like this, in addition to those that are well recognised, can inform on which stakeholders may be disenfranchised or marginalised, but may also provide insight into disparities between an academic view of stakeholder networks versus what key stakeholder groups experience on the ground. If marginalisation was identified during initial stakeholder mapping exercises, focusing on opening two-way dialogue with stakeholders who would otherwise be considered peripheral would benefit the stakeholder analysis (Hart & Sharma 2004). In this case, however, stakeholders that are often identified as high influence and high interest, or 'key players' in stakeholder mapping exercises were omitted by participants (Reed *et al.* 2009). Due to the UK Department for Environment Food and Rural Affairs often closely liaising with the Scottish Government, they were expected to be perceived as relatively influential. The lack of responses for the EU Governance structure were particularly counter-intuitive given their role in proposing and administering relevant legislation relating to land and water management, such as the EU Water Framework Directive, the EU Bathing Water Directive, the EU Floods Directive and EU Climate Change targets.

The lack of responses may be methodological. When asked which stakeholders had influence on catchment management in their respective catchments, this may have implied influence would have to be through direct actors, rather than indirectly through legislation. The Environmental Regulator implements most of the EU Directives and is the designated competent authority; hence stakeholders in the catchment may perceive them as the stakeholder that influences their catchments through regulation and enforcement. Had the methodology encouraged more hierarchical thinking in participants, the UK Government and EU Commission would likely have been mentioned more often; however, such a methodology may have overly focused on participants' knowledge of the stakeholder network rather than allowed insight into their day-to-day experiences within their catchments. When engaging participants for stakeholder mapping, surveys also often make use of a pre-prepared list of stakeholder groups to choose from, which does not supply the information of who might have been left out (Zingraff-Hamed *et al.* 2020). Arguably, our methodology is an approach to understand the procedural interactions among land and water stakeholders, which may show that these higher level institutions are at arm's length when it comes to catchment management in practice. Stakeholder analysis in the Swiss water supply and wastewater sector showed clear dominance of local actors, while regional and especially national actors were perceived as less important (Lienert *et al*. 2013). Including Scottish Government staff as participants would have likely included the UK Government and EU Commission in the responses as they would have more direct links due to their closer proximity on the stakeholder flow chart in Figure 5.1. When selecting stakeholders as participants for a participatory exercise, a line needs to be drawn at some point to decide who to involve, so it is effectively a chicken-and-egg situation. The composition of participants will influence results, so initial selection of participants needs to have a clear aim in mind. In our example, we focused on participants directly involved in land and water management and chose not to include legislators.

5.4.4. Degree and core and periphery analysis

Stakeholders that had the largest degree score and which formed the core of the network were from across all sectors, thus showing no bias towards any specific sector. The stakeholder nodes that were classed as being within the core across all four interviewed stakeholder typologies and across all three study catchments may be assumed as particularly relevant for catchment management across the country (Environment Agency, Local Government, River Trusts and the Timber Industry). Key players in these elicited stakeholder networks included legislators and land managers similarly to Reed *et al.* (2009) in their example of a stakeholder mapping exercise applied to flooding. Our results also included River Trusts, public bodies, agencies and private businesses, which may illustrate the heterogeneity and complexity of focusing on integrated catchment management networks as opposed to a single water management issue. Integrating peripheral stakeholders into participatory catchment management can help to achieve more equitable development outcomes where people are marginalised, but may also aid behaviour change of badly networked stakeholders, which may be of particular interest if they are likely to participate in illegal behaviours (de Lange *et al.* 2019).

Numbers of times stakeholders were mentioned and co-occurrence of stakeholders were used here to determine nodes and ties in the one-mode networks. Although social network analysis would allow a whole host of other social network analysis tools, this would also require a much more rigorous data collection to ensure each node and tie in the network was identified (Lienert *et al.* 2013). Hence our methodology has the benefits of a rapid survey with broad but still in-depth insights into stakeholder networks. The methodology could be adapted to include elicitation of hierarchical level of stakeholders or interest as well as influence to make it more comprehensive. Other additional criteria, such as identifying stakeholder roles, may also be beneficially added to techniques such as the one presented here (Heidrich *et al.* 2009). Alternatively this methodology could be used as a complementary tool for enhancing other stakeholder mapping techniques or as a preliminary methodology for full social network analysis. For example, the methodology could be used as a quick scoping exercise, with benefits exceeding a simple desktop study, to identify which stakeholders to involve in any stakeholder participation exercise. On the other hand, it could provide a framework to help catchment co-ordinators identify likely stakeholders of interest for catchments of a particular typology without the need for further elicitation or participation. Increased understanding of stakeholder networks can improve stakeholder communication, make implementation more effective and make citizen science initiatives are more successful (Skarlatidou *et al.* 2019). It could inform who to target for consultation in 'active involvement' exercises suggested under the WFD or determine who to involve as stakeholders in a Citizens' Jury to inform decision-making on complex science-policy problems (Fish *et al.* 2013).

5.5. Conclusion

Combining stakeholder analysis and social network methods provides a novel and quick tool to investigate stakeholder interactions, in our case concerning catchment management. Comparing outputs from the analysis of three contrasting river catchments as well as between participants from four key stakeholder groups allowed identification of which stakeholders are more central to the catchment management networks as opposed to which are seen as to act more along the periphery and to quantify perceived stakeholder influence on the water environment as well as whether that influence was perceived to be mostly positive, neutral or negative.

Social network analysis into how information flows between the core and peripheral stakeholder groups identified here may help provide more effective communication within Scottish land and water management stakeholder networks. For example, communications may be targeted to highly connected opinion leaders to leverage their influence, or communication may be facilitated between distinct subgroups to promote peer learning (de Lange *et al.* 2019). Future research into social networks could test the hypothesis whether a more centralised network structure may be more effective at coordinating catchment management, or whether relying on a single dominant node within the core may lead to conflicts and lack of cooperation between other nodes. Our catchments seem to show a mixture of those two approaches as the Environmental Regulator dominated, but the other core nodes were also well-connected, which may give them both the advantages of being able for rapid top-down mitigation of specific and easily identifiable threats to the system, whereas broader connectivity may allow more effective management of other more indirect or less easily measurable threats (Nuno *et al.* 2014).

Applying ‘fully articulated’ social-ecological network analysis to a catchment socio-ecological system presents an innovative avenue to further investigate not just ties between social network nodes, but also relationships in ecological networks and social-ecological ties (Sayles *et al.* 2019). Such research could give insights into how collaboration among users of shared catchment resources leads to successful management (Bodin 2017), or which social-ecological patterns are likely to facilitate adaptations and transformations (Barnes *et al.* 2017). The social factors that provide resilience within catchment management, such as flexibility, social organization, learning, and agency could also be explored with social-ecological network approaches (Cinner & Barnes 2019). This may give vital insights into the social dimensions of resilience in catchments to help understand likely impacts of land management changes or climate change and help to make catchments more resilient against future change.

Expert and non-expert preferences for managing landscapes to deliver multiple benefits from agri-environment measures

6.6. Introduction

The Green Revolution was one of the most important technological transformations of the 20th century. It dramatically improved crop yields, surpassing the demand of a growing global population, hence substantially reducing infant mortality in the developing world, saving an estimated 100 million infants from death of starvation over four decades (Goltz *et al.* 2020). While such a focus on increasing yields has alleviated hunger and poverty, it has neglected consideration of nutritional and environmental dimensions of food production (Davis *et al.* 2019). Negative side-effects of the intensification of agriculture include diminishing freshwater resources, soil, air and water pollution, soil erosion, loss of soil fertility, decrease in nutrient density of crops and extinction of traditional cultivars (John & Babu 2021). A significant research challenge for post-Green Revolution farming is how to provide food security while also minimising environmental impacts and protecting human health and well-being. Arguably at the crux of achieving sustainable global food systems is our ability to incorporate mitigation planning and implementation to maximise ecosystem service provisioning of farmland (Davis *et al.* 2019). As rivers integrate pressure from across their catchments, they are an appropriate scale for assessing land and water management issues. Globally, water quality objectives are still not being met in many countries, and eutrophication problems like harmful algal blooms and hypoxia are increasing in many developing regions such as Southeast Asia (Strokal *et al.* 2016). Water quality remains a key driver of land and water management, but to maximize cross-sectoral societal objectives, it must be combined with functions for biodiversity, flooding and climate adaptation as part of more integrated catchment management (Stutter *et al.* 2019).

In the European Union (EU), agri-environment measures (AEMs) are designed to partly address the detrimental environmental impacts of agriculture. They present a majority of the conservation expenditure in Europe, but it is disputed to what extent they are able to enhance biodiversity and protect aquatic environments from agricultural pollution (Batáry *et al.* 2015; Randall *et al.* 2015; Vesterager *et al.* 2012). Results on the effectiveness of AEM are often contradictory as they accommodate much variability due to a diverse range of AEM, differences in the design of AEM, differences in the general characteristics of agroecosystems, and spatial and temporal landscape heterogeneity (Manning *et al.* 2018). Ineffective spatial targeting and management of AEMs wastes financial resources by paying farmers to implement mitigation measures where they may fail to deliver the maximum environmental benefits (Psaltopoulos *et al.* 2017). Field trials have evaluated the efficacy of mitigation measures deployed in isolation in localised experiments; however, the evidence base to understand the relative efficacy of multiple mitigation measures deployed at the catchment scale accommodates more uncertainty, lacking the degree of control and manipulation associated with more reductionist field-trial settings (Jones et al. 2013). Furthermore, water quality improvements may take decades to become detectable at larger landscape scales because of lag effects from legacy pollution (Melland *et al.* 2018). Uncertainties are compounded when trying to select mitigation measures for the delivery of multiple benefits. Increasingly, modelling is used to explore the effects of landscape management options simultaneously, providing insight on how best to balance ecosystem service trade-offs when dealing with multiple environmental objectives (Cord *et al.* 2017; Verhagen *et al.* 2018; Zhang *et al.* 2017).

Using expert knowledge to investigate causal links between mitigation measures and ecosystem service delivery is a way to reduce the costs of monitoring (Reed *et al.* 2014). A range of research methodologies have been used to elicit expert opinion on environmental issues. Qualitative approaches such as interviews, diary-based research, and facilitated group discussion at workshops or via focus groups and expert panels can help to determine individual or group views (Finn *et al.* 2009). Alternative semi-quantitative approaches include the use of Likert and other rating scales within surveys or discrete choice experiments to analyse choice behaviour and determine preference scores for alternative items (Cross *et al.* 2012; Mills *et al.* 2021). Best Worst Scaling (BWS) is a discrete choice technique that enables large numbers of stand-alone items to be ranked. It was developed for market research but has been widely employed in health and social science research (Erdem *et al.* 2012; Farkas *et al.* 2021; Lagerkvist *et al.* 2012; Muunda *et al.* 2021). Recently, BWS has also been successfully used to evaluate mitigation measures in environmental management for forest management (Loureiro & Dominguez Arcos 2012), and greenhouse gas mitigation in farming (Jones *et al.* 2013), as well as for evaluating ecosystem services of urban forests (Soto *et al.* 2018), energy policy trade-offs (Aruga *et al.* 2021), bioenergy generation from municipal waste (Alidoosti *et al.* 2021), water resource management (Ma *et al.* 2021), and ecosystem and economic benefits of water body restoration and conservation (Tyner & Boyer 2020).

In this context, we used a mixed method survey that included BWS alongside multiple choice, Likert-scale, open-ended and ranking questions, to elicit expert views on the effectiveness of AEMs with respect to their ability to reduce diffuse pollution on downstream water courses, increase habitat quality to support biodiversity and attenuate flood waters. The objectives of the study were to (1) evaluate the perceived relative effectiveness of ten AEMs to achieve multiple benefits, (2) quantify differences in views on AEM effectiveness between expert and non-expert stakeholders, and (3) determine the opportunities and challenges of using AEMs for multiple ecosystem service benefits in Scottish catchments.

6.7. Methodology

6.2.1. Survey design and data collection

We designed an online survey for distribution to stakeholders with expertise in land and water management in Scotland, with the survey period starting in February 2019. Participants were identified initially through a desk-based exercise (using search terms “diffuse pollution”, “water quality”, “biodiversity”, “flooding, natural flood management”, “river”, “catchment”), with additional experts identified via recommendations from initial participants, i.e. a snowballing approach. The survey was designed and implemented using Sawtooth SSI Web Survey software version 7 (Sawtooth Software Inc. 2007). Emails were sent to 221 potential participants in Scotland and a flyer was distributed in research and higher education institutes (Appendix 1), which outlined the rationale of the engagement exercise and the link and QR code to the online survey. To maximise survey completion rates, the questionnaire was relatively short (ten questions plus three sets of BWS exercises) and did not require the participants to give any personal details. The survey included a brief introduction providing details about the project, an assurance of confidentiality and a consent form (see Annex 2 for complete survey). The first five questions inquired about the participant's stakeholder group (e.g. farmer or researcher), job role, experience working in Scottish river catchments and whether they had the greatest expertise on how land management impacts either biodiversity, diffuse pollution, or flooding. This was followed by three sets of BWS exercises. Following completion of the three BWS exercises, participants were asked how common they perceived the uptake of the 10 AEMs in Scotland to be and to what extent their implementation should be increased throughout Scotland. Next, they were asked to rank the 10 AEMs by their perceived overall benefit. Finally, participants were asked two open-ended questions, where they could share their views on (i) which other on-farm measures not considered in our study would support multiple benefits; and (ii) the benefits and shortfalls of agri-environment measures and what limits their uptake in Scotland. The responses to the open-ended questions were counted and grouped into themes according to content.

BWS, also known as Maximum Difference Scaling, allows the ranking of larger lists of items by presenting participants with a smaller subset of the options and asks them to choose the two items within each set that they consider the 'best' and 'worst' options for a particular scenario, or the least or most preferred option (or important, useful, causing most concern, etc.). The pair of items chosen as best/worst exhibits the maximum difference in preference. BWS avoids ambiguity in comparison to approaches such as Likert scaling where multiple options may be selected as having the greatest or lowest scores (Louviere *et al.* 2013). BWS surveys also reduce the likelihood of results being biased due to acquiescence and/or extreme response bias. Acquiescence bias refers to a tendency to agree with items on questionnaires, while extreme response bias refers to a tendency to choose items on the ends of a scale (Shoji *et al.* 2021). Compared to rating and ranking exercises, BWS allows for more accurate judgements of comparative values, especially when presented either with a long list of items or when the choices incorporate cognitively demanding choices (Lee *et al.* 2008). BWS surveys perform better than standard rating scales not just in terms of discriminating among items but also in discriminating among respondents on the items (Orme & Chrzan 2006). BWS is also an efficient mechanism for capturing a large amount of information per unit of respondent effort (Orme 2010). Taking the example of a BWS exercise with four items (A, B, C and D), by identifying A as best and D as worst, the respondent identifies A preferrable to B, C and D and A, B and C preferable to D. The only unknown is how B compares to C. Thus, with just two clicks, the respondent has provided information regarding five of the six possible paired comparisons within the set.

The ten AEMs used in our study (Table 6.1) were selected from 18 AEMs, focused on protecting water resources, available for Scottish farms through the European Union 2018 Rural Development Agri-Environment and Climate Scheme. These were then scored by two members of the research team on whether they, improving water quality, had potential to provide additional environmental benefits by also (1) increasing biodiversity, (2) attenuating flooding, (3) reducing diffuse pollution, (4) mitigating climate change and/or (5) enhancing landscape features. Each AEM could score a maximum of ten points and the ten highest scoring AEMs were selected for the survey, with the exception of one AEM ('Stubbles followed by green manure in arable rotation') which was excluded as it was deemed too similar to one of the other highest scoring AEM ('Retain winter stubbles until early spring').

Table 6.1: The ten AEM options used in generating BWS choice cards.

No.	AEM option
1	Establish and maintain rural sustainable drainage systems.
2	Convert arable land at risk for erosion or flooding to low-input grassland.
3	Create and manage hedgerows.
4	Provide alternative drinking water sources for livestock.
5	Restore and protect riverbank vegetation damaged by historic grazing and poaching.
6	Remove, lower or breach embankments to restore floodplains.
7	Retain winter stubbles until early spring.
8	Establish water margins (minimum width 3, 6 or 12 metres for burns, streams and lochs).
9	Reduce sheep stocking rates on moorlands.
10	Establish grass strips within or at the edges of arable fields (minimum width 3 metres).

As catchments are highly variable in geology, topography, land cover and land management, participants were asked to complete the BWS exercises not based on a specific catchment, but instead to consider the benefits of AEMs if they were implemented across farms in a 'model Scottish catchment', with a mixture of typical, current farming practices. Land use in this model catchment was therefore specified to be broadly in line with agricultural production across Scotland as a whole consisting of 30% upland rough grazing, 15% lowland improved grazing and 10% arable agriculture common in Scotland such as growing barley, potatoes and oilseed rape.

In our survey, a total of ten AEMs were shown to the experts in subsets depicting four AEMs at a time. Four or five items per set are regarded as optimal for respondent evaluation, more than this may lead to respondent fatigue (Sawtooth Software, 2007). This choice task was then repeated ten times with different combinations of AEMs. To keep the number of AEMs per set to a manageable four options and ensure each AEM was shown to each respondent 3-5 times, we used ten choice sets per BWS exercise. A total of 30 sets were shown to participants to elicit their views on the likely effectiveness of the ten different AEMs to (a) decrease diffuse pollution to downstream water courses, (b) increase habitat quality to support biodiversity and (c) attenuate flood waters (Figure 6.1).

Reducing Diffuse Pollution

Please picture a model Scottish catchment with a land cover of 30% rough grazing, 15% improved grazing and 10% arable agriculture.

Considering only the following 4 measures, which do you think is most likely and which is least likely to reduce diffuse pollution from farm land to downstream water courses, assuming they were utilised widely in the catchment (in around 50% of all farms)?

(9 of 10)

	most likely	least likely
Convert arable land at risk for erosion or flooding to low-input grassland.		
Establish and maintain rural sustainable drainage systems.		
Remove, lower or breach embankments to restore floodplains.		
Establish water margins (minimum width 3, 6 or 12 metres for burns, streams and lochs).		

Figure 6.1: An example of a choice set from one of the best–worst scaling exercises.

6.2.2. Statistical analysis

The BWS data were analysed through counting analysis and hierarchical Bayes (HB) score estimation. In the counting analysis, to combine the information from best and worst judgements, the proportion of times an option was selected as worst was subtracted from the proportion of times it was chosen as best. The experimental design of BWS surveys ensures that each option is shown an equal amount of time, allowing the estimation of the relative preference (or "utility") of each of the options. HB analysis not only provides more stable and accurate utility scores for each AEM, but also utility scores for each AEM from every individual respondents – called part-worth utilities. HB involves applying a statistical model using thousands of iterations to estimate part-worth utilities. It runs a separate lower- and upper-level model for the data, hence the term "hierarchical". In the lower-level model, HB considers how well part-worth utilities fit each respondent's choices while the upper-level model estimates overall part-worth utility averages and variances for the sample population, including the covariances between part-worths across the respondents. This improves individual respondent estimation as it borrows information from other respondents in the sample to stabilize the estimates for each individual. The more consistent respondents are in their responses, less information is taken from the population characteristics. In less consistent or atypical respondents, more information is taken from population characteristics to move them toward the mean and stabilise their part-worths (Orme 2010). Here we used both the 'best' and the 'worst' choices to inform HB and used 30 000 total iterations of the model, 20 000 iterations before using results and 10 000 draws for each respondent. The HB probabilities for diffuse pollution, biodiversity and flood mitigation responses were summed to compare the collated results to the ranking of the AEMs from question 8, where the 10 AEMs were ranked by which had the greatest overall environmental benefits. Both the collated HB probabilities and the mean rank from question 8 were scaled to values from 0 to 1 by dividing by the maximum mean.

Root likelihood (RLH) is a measure of fit between utility scores and choices made by respondents to BWS surveys and refers to how consistent respondents are in applying an evaluative strategy and assigning ratings or choices (Orme 2010). Using utility estimates, one can predict the likelihood that respondents would make the choices that they actually made within each choice task. RLH is the geometric mean of estimated probabilities associated with the alternatives actually chosen by respondents, or the n^{th} root of the likelihood.

Basic descriptive statistics for means, variances and distribution analysis of scores were obtained and statistical analysis undertaken using SPSS version 28 (IBM 2021). To compare responses for different AEMs, a non-parametric statistical test (Kruskall-Wallis) was used, as variances were often significantly different per Levene's homogeneity of variances test and some of the BWS scores were not normally distributed (AEM 4 in the biodiversity BWS exercise and AEM 1, 4 and 6 in the flood mitigation BWS exercise). To estimate which particular AEMs were statistically different from each other a James Howell post-hoc test was carried out, due to unequal variances. A significance threshold at the 0.05 level was used for all statistical tests.

Expert stakeholders in this study were defined as those that self-identify as having the greatest expertise in one of the three options from question 5 – diffuse pollution, biodiversity or flooding impacts of land management. Thus, each of the participants was classed as an expert in one of the three BWS exercises and a non-expert in the other two that considered scenarios outside of their core field of interest. Responses from expert and non-expert stakeholders were compared using a non-parametric statistical test (Mann-Whitney) as the assumptions of normal distribution, equal variances and equal sample size were not met. The Scheirer-Ray-Hare extension to the Kruskall-Wallis test was used to analyse the effects of the interaction between the types of AEM and whether participants were from the expert or non-expert group. In SPSS, the scores from the three BWS exercises were ranked and replaced by numbers from 1 to n. A two-way ANOVA was then carried out using the ranked ordinal data and the corresponding AEM and expert group.

6.8. Results

A total of 68 stakeholders (31% response rate to invitations) participated in the study, although only 54 completed the entire online survey and three sets of BWS questions. When combined, participants scored a total of 1674 BWS sets – 584 on diffuse pollution, 550 on biodiversity and 540 on flood mitigaton benefits of the ten AEM (Table 6.2). The participants identified as belonging to one of nine possible stakeholder groups: Scientists (20), Environmental Regulators (14), Fisheries Trusts (7), Farm Advisors (6), National Park Authority (3), Farmers (2), Local Authority (2), Water Industry (2) and "Other" (12). Most stakeholders had more than 10 years experience working in their job role (40), followed by by 5-10 years (12), 1-2 years (6), 2-5 years (5) and less than one year (3). Despite an extra round of participant selection only about half as many stakeholders identified their greatest area of expertise as relating to flooding (13) as compared to biodiversity (26) or diffuse pollution (27).

Table 6.2: BWS counts analysis for diffuse pollution, biodiversity and flood mitigation BWS exercises.

Item Number	Times Shown	Times Selected Best	Best Count Proportion	Times Selected Worst	Worst Count Proportion	BW proportion difference
AEM scored by their likely ability to reduce diffuse pollution in the model catchment n = 68, Total number of BWS sets = 584						
1. Rural Sustainable Drainage Systems	233	60	0.258	36	0.155	0.103
2. Arable land to grassland conversion	238	93	0.391	21	0.088	0.303
3. Create and manage hedgerows	232	8	0.034	150	0.647	-0.613
4. Alternative water sources for livestock	232	86	0.371	32	0.138	0.233
5. Reduce poaching of river bank	235	88	0.374	24	0.102	0.272
6. Restore floodplains	231	23	0.1	109	0.472	-0.372
7. Retain winter stubbles	233	32	0.137	43	0.185	-0.048
8. Riparian buffer zones	229	144	0.629	5	0.022	0.607
9. Reduce moorland sheep density	235	11	0.047	126	0.536	-0.489
10. Arable field grass buffer zones	238	39	0.164	38	0.16	0.004
AEM scored by their likely ability to increase bidiversity in the model catchment n = 55, Total number of BWS sets = 550						
Item Number	Times Shown	Times Selected Best	Best Count Proportion	Times Selected Worst	Worst Count Proportion	BW proportion difference
1. Rural Sustainable Drainage Systems	220	16	0.073	96	0.436	-0.363
2. Arable land to grassland conversion	220	46	0.209	50	0.227	-0.018
3. Create and manage hedgerows	220	109	0.495	16	0.073	0.422
4. Alternative water sources for livestock	220	9	0.041	141	0.641	-0.6
5. Reduce poaching of river bank	220	71	0.323	13	0.059	0.264
6. Restore floodplains	220	65	0.295	46	0.209	0.086
7. Retain winter stubbles	220	23	0.105	85	0.386	-0.281
8. Riparian buffer zones	220	120	0.545	4	0.018	0.527
9. Reduce moorland sheep density	220	42	0.191	72	0.327	-0.136
10. Arable field grass buffer zones	220	49	0.223	27	0.123	0.1
AEM scored by their likely ability to improve flood mitigation in the model catchment n = 54, Total number of BWS sets = 540						
Item Number	Times Shown	Times Selected Best	Best Count Proportion	Times Selected Worst	Worst Count Proportion	BW proportion difference
1. Rural Sustainable Drainage Systems	216	101	0.468	19	0.088	0.38
2. Arable land to grassland conversion	216	64	0.296	25	0.116	0.18
3. Create and manage hedgerows	216	14	0.065	77	0.356	-0.291
4. Alternative water sources for livestock	216	5	0.023	160	0.741	-0.718
5. Reduce poaching of river bank	216	56	0.259	30	0.139	0.12
6. Restore floodplains	216	170	0.787	6	0.028	0.759
7. Retain winter stubbles	216	19	0.088	65	0.301	-0.213
8. Riparian buffer zones	216	62	0.287	15	0.069	0.218
9. Reduce moorland sheep density	216	30	0.139	102	0.472	-0.333
10. Arable field grass buffer zones	216	19	0.088	41	0.19	-0.102

The AEMs perceived to be most common in Scotland were 'Riparian buffer zones' (52% common or very common; Figure 6.2), 'Create and manage hedgerows (51%), and 'Alternative water sources for livestock' (49%). 'Restore floodplains' (2%), 'Arable land to grassland conversion' (2%) and 'Rural sustainable drainage systems' (9%) were scored as being the least common.

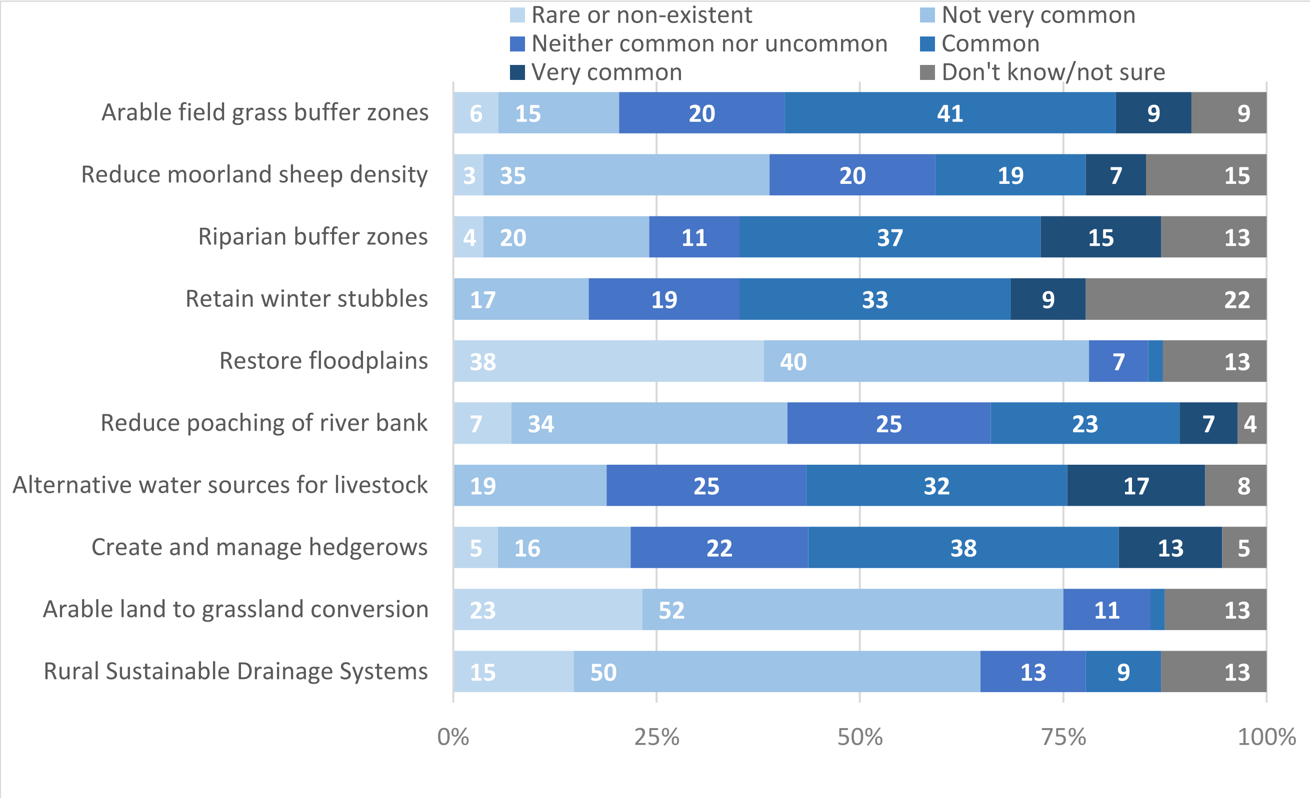


Figure 6.2: Percentage of the stakeholders that perceived the ten AEM as rare or non-existent (lightest blue), not very common (light blue), neither common nor uncommon (mid-blue), common (darker blue), very common (darkest blue), Don't know/not sure(grey); n=68.

AEMs that were most often strongly supported by stakeholders were ‘Riparian buffer zones’ (61%) ‘Restore and protect riverbank vegetation damaged by historic grazing and poaching’ (59%), ‘Alternative drinking water sources for livestock’ (57%). The lowest support was for ‘Reducing moorland sheep density’ (Figure 6.3).

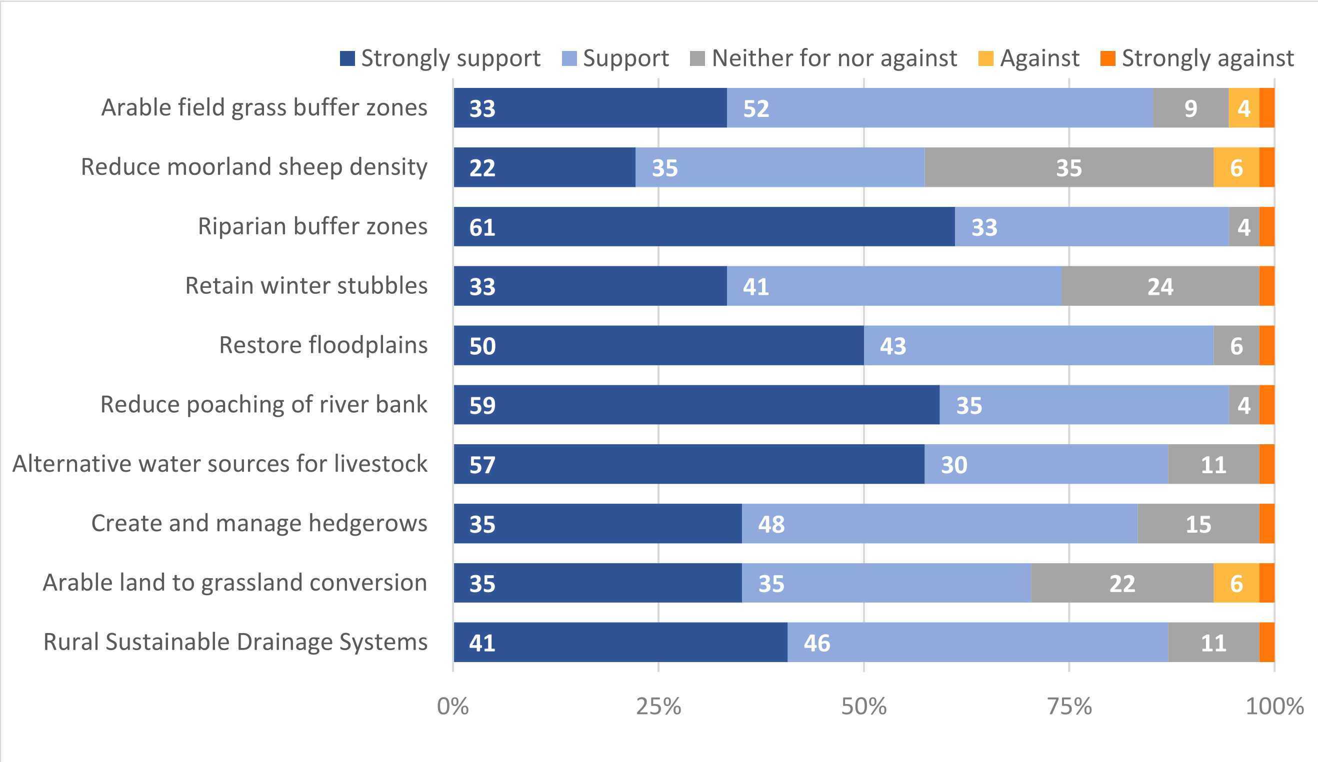


Figure 6.3: Stakeholder perception as to whether the implementation of the ten AEM should be increased across Scottish farms: Strongly support (dark blue), support (light blue), neither for nor against (grey), against (amber), strongly against (orange) ; n=68.

The 68 participants suggested 117 other on-farm measures not considered in the study to support multiple benefits in response to question 9. ‘Planting and managing trees and scrub on farms’ (i.e. upland woodland creation, wet woodland planting, agroforestry, shelterbelts etc.) was most common (35 participants), with riparian zone tree planting being a particularly dominant response (12). This was followed by 27 participants mentioning ‘Improved field management’ (i.e. ploughing direction, reducing artificial inputs (fertiliser/pesticides), ground cover etc.), Peatland/moorland restoration (9), and wetland creation (9). On-farm biodiversity measures such as species-rich grassland and wildlife corridors were mentioned eight times, and in-stream biodiversity and river restoration measures such as re-meandering, introducing large woody debris and flood-bank removal were mentioned seven times.

Results from the HB analysis illustrate how stakeholders scored AEMs differently for their benefits to water quality, biodiversity and flood mitigation (Figure 6.4). ‘Restoring floodplains’ was viewed as the best option for reducing flooding and ‘riparian buffer zones’ scored highest for diffuse pollution and biodiversity improvements, as well as moderate flood water alleviation.

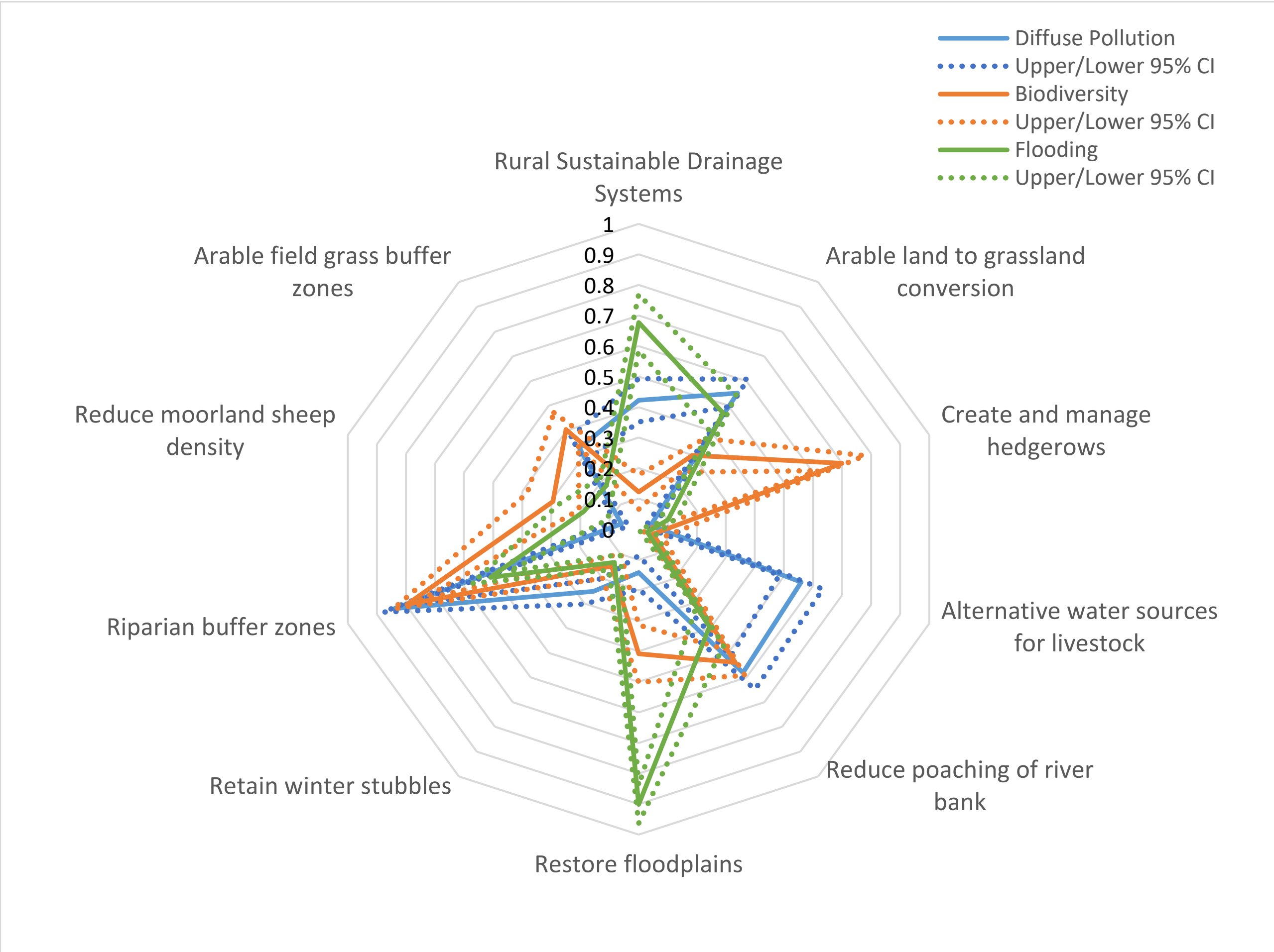


Figure 6.4: Hierachichal bayes probability score of BWS responses for likely benefits of the 10 AEM to improve diffuse pollution (blue), biodiversity (amber) and flood mitigation (green) across the model catchment. Dotted lines indicate 95% upper and lower confidence intervals. The RLH of the HB modelling was 0.58 for the diffuse pollution BWS exercise, 0.59 for the biodiversity part and 0.64 for the flood mitigation exercise.

When combining the three sets of BWS scores, ‘riparian buffer zones’ scored the highest for delivering multiple benefits (Figure 6.5, black lines). The ranking exercise (Question 8) confirmed ‘riparian buffer zones’ as the AEM perceived as having the greatest overall ecosystem benefits and ‘reducing poaching of the river bank’ and ‘restoring floodplains’ were ranked as second and third highest overall, respectively. The RLH for the diffuse pollution BWS set was 0.60 ± 0.02 , 0.61 ± 0.02 for the biodiversity sets and 0.65 ± 0.02 for the flood mitigation ones. The 10 AEMs were significantly different from one another in all of the three BWS exercises (Table 6.3), and post-hoc analysis showed that the majority of the AEMs were distinctly different from each other (Table 6.4).

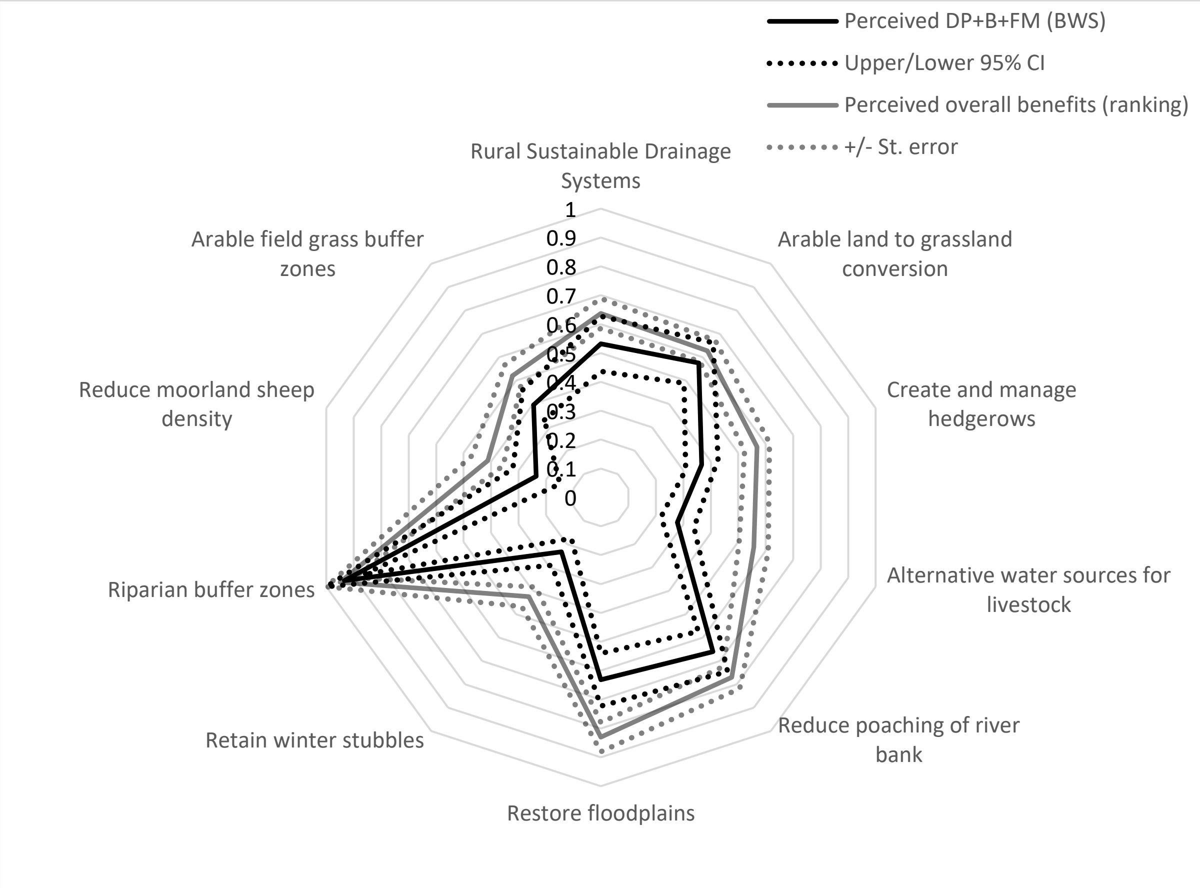


Figure 6.5: Comparing the cumulative multiple benefits from the three BWS exercises (diffuse pollution + biodiversity + flood mitigation (DP + B + FM) black lines with 95% confidence intervals and the mean scores from the ranking exercise from question 8 – perceived greatest overall environmental benefits (grey lines ± standard errors (Q8). Both data sets have been normalised (min-max) to allow direct comparison.

Table 6.3: Kruskall-Wallis test results comparing the medians of the utility scores each of the ten AEMs received in the three BWS exercises.

	Water quality	Biodiversity	Flood mitigation
Kruskall-Wallis H	**411.84**	**284.16**	**333.08**
n	68*10 = 680	61*10 = 610	54*10 = 540
Asymp. Sig.	**<.001**	**<.001**	**<.001**
	Kruskall-Wallis test Mean Ranks		
1. Rural Sustainable Drainage Systems	385.10	168.56	398.94
2. Arable land to grassland conversion	457.01	285.07	339.13
3. Create and manage hedgerows	101.10	462.57	156.80
4. Alternative water sources for livestock	448.44	108.16	62.06
5. Reduce poaching of river bank	464.66	399.89	309.81
6. Restore floodplains	182.40	329.64	483.54
7. Retain winter stubbles	295.16	196.28	195.48
8. Riparian buffer zones	592.94	500.67	360.67
9. Reduce moorland sheep density	133.19	267.69	169.74
10. Arable field grass buffer zones	344.99	336.48	228.83

No significant differences were found between the part-worth utility medians of expert and non-expert stakeholders in water quality BWS sets (p = 0.752, U = 0.100, n = 630), biodiversity BWS sets (p = 0.178, U = 0.256, n = 610), or flood mitigation BWS sets (p = 0.825, U = 0.049, n = 540). The Scheirer-Ray-Hare extension to the Kruskall-Wallis test showed no significant effect of the interaction between the types of AEM and whether participants were from the expert or non-expert group. Mean part-worth utilities from expert stakeholders and non-expert stakeholders only statistically significantly differed between AEM number 4 (‘provide alternative drinking water sources for livestock’) in the diffuse pollution BWS responses (p = 0.046, U = 3.985, n = 68, Figure 6.6).

Table 6.4: Games Howell post-hoc test results for the HB utility scores from the diffuse pollution, biodiversity and flood mitigation BWS exercises. Significance of differences between the means at 0.05 level.

Diffuse pollution										
	1	2	3	4	5	6	7	8	9	10
1										
2	.097									
3	<.001	<.001								
4	.529	1.000	<.001							
5	.051	1.000	<.001	1.000						
6	<.001	<.001	.025	<.001	<.001					
7	.008	<.001	<.001	<.001	<.001	<.001				
8	<.001	<.001	<.001	<.001	<.001	<.001	<.001			
9	<.001	<.001	.231	<.001	<.001	.887	<.001	<.001		
10	.667	<.001	<.001	.004	<.001	<.001	<.001	<.001	<.001	
Biodiversity										
	1	2	3	4*	5	6	7	8	9	10
1										
2	<.001									
3	<.001	<.001								
4*	.157	<.001	<.001							
5	<.001	<.001	.011	<.001						
6	<.001	.813	<.001	<.001	.308					
7	.830	.002	<.001	<.001	<.001	<.001				
8	<.001	<.001	.929	<.001	<.001	<.001	<.001			
9	.057	.997	<.001	<.001	.001	.579	.544	<.001		
10	<.001	.323	<.001	<.001	.025	1.000	<.001	<.001	.301	
Flood mitigation										
	1*	2	3	4*	5	6*	7	8	9	10
1*										
2	.003									
3	<.001	<.001								
4*	<.001	<.001	<.001							
5	<.001	.912	<.001	<.001						
6*	<.001	<.001	<.001	<.001	<.001					
7	<.001	<.001	.152	<.001	<.001	<.001				
8	.020	.983	<.001	<.001	.228	<.001	<.001			
9	<.001	<.001	1.000	<.001	<.001	<.001	.625	<.001		
10	<.001	<.001	<.001	<.001	<.001	<.001	.381	<.001	.079	

*data was not normally distributed

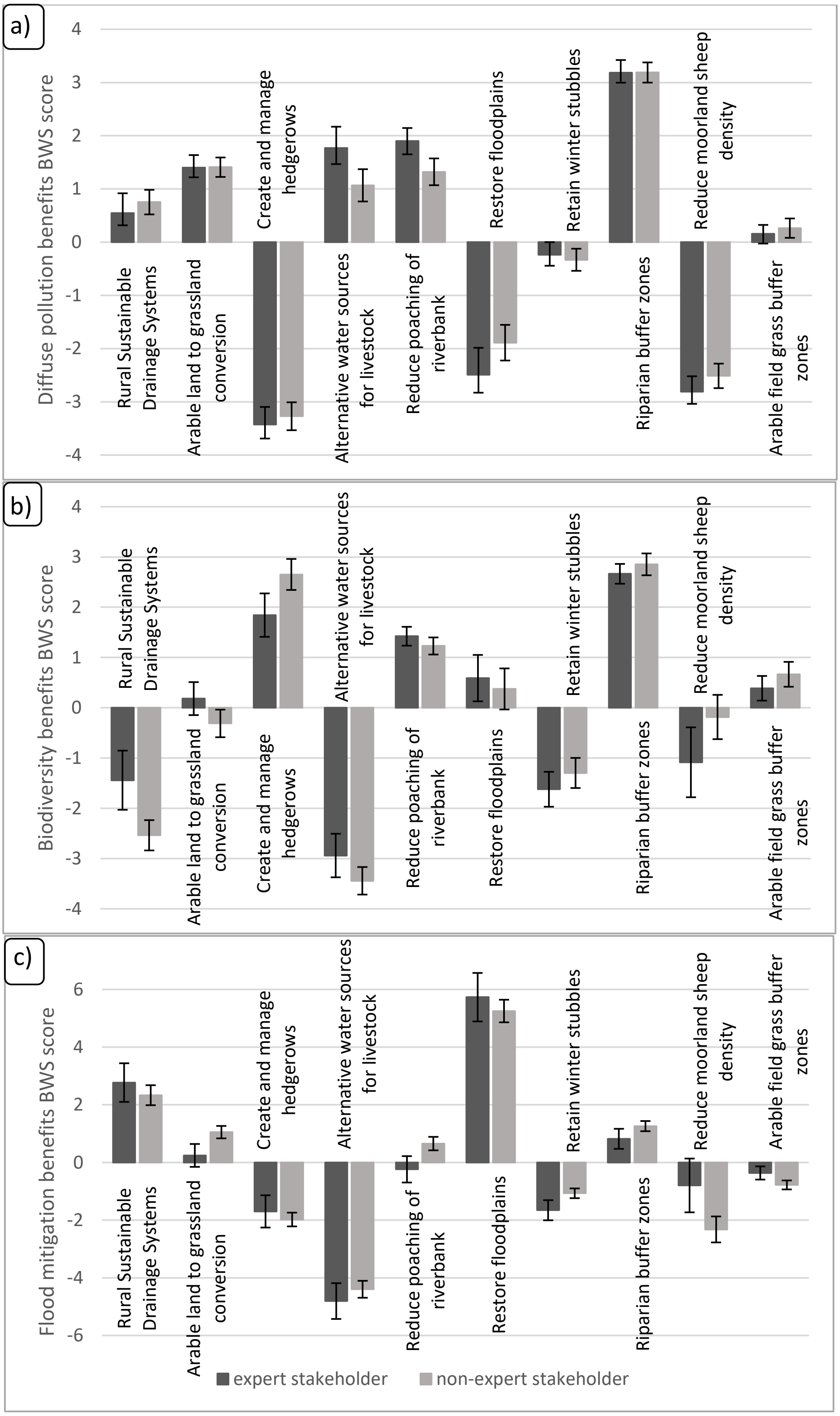


Figure 6.6a, b & c: Mean BWS scores of the perceived benefits the ten AEM regarding their ability to reduce diffuse pollution (a, n = 26 expert, 42 non-expert), increase biodiversity (b, n = 22 expert, 33 non'expert') (b) and mitigate flooding (c, n = 10 expert, 44 non-expert). Establish and maintain rural sustainable drainage systems.

6.9. Discussion

The multi-method approach presented here elicited expert views on the likely landscape-scale effectiveness of AEMs with respect to their ability to reduce diffuse pollution of downstream water courses, increase habitat quality to support biodiversity and attenuate flood waters in Scottish catchments. To our knowledge, BWS has not previously been used for assessing expert views on the effectiveness of AEMs to improve ecosystem service provisioning. The methodology allowed distinct evaluation of the perceived relative effectiveness of ten AEMs to achieve multiple benefits, with 'riparian buffer zones' receiving the greatest overall scores for delivering environmental win-wins. Catchment managers often deal with spatial and temporal variability, the existence of thresholds and 'tipping-points' in catchment systems, non-linearity and the potential for irreversible collapse of ecosystems when making decisions at the landscape scale (Gain *et al.* 2021). With inherent uncertainty concerning the best 'mitigation mix' to adopt at the landscape scale in order to achieve multiple win-wins, it is essential that those with a responsibility for decision-making have access to the best-available expert knowledge (Jarvie *et al.* 2013). The BWS approach presented here can help to inform decision-making under uncertainty, providing access to expert knowledge to help guide policy implementation in complex catchment systems (Knol *et al.* 2010). While there remains uncertainty over how effective AEMs might be when deployed at the catchment scale and under different scenarios, this should not paralyse mitigation planning and implementation. Instead, we need mechanisms that allow us to capitalise on the best available expert knowledge to help prioritise and promote the use of AEMs, and BWS offers such an opportunity.

Our use of a BWS approach to evaluate landscape management options could be used for other 'model' catchments as part of scenario analysis, e.g., to explore how different catchment characteristics influence AEM preferences among experts. Likewise, BWS could be deployed to determine expert views on AEM performance for actual catchments across the UK and beyond, where known issues are currently under debate. In the UK many catchment partnerships exist that are designed to achieve multiple environmental objectives and there is interest in how to integrate different objectives to meet the needs of multiple policy drivers, e.g., the WFD and Floods Directive. Thus, the approach could be used as a participatory stage, e.g. via focus groups or workshops using the Delphi method, for helping to facilitate AEM selection when designing agri-environment programmes at a national level, or used more locally to inform the decision-making of land managers in light-touch regulated countries, such as New Zealand, where there are no AEM subsidies available (Knook *et al.* 2020).

On the whole, expert and non-expert stakeholders scored the importance of different AEMs similarly across all the BWS exercises, which likely reflects the breadth of knowledge that land and water management stakeholders posess. This was also evident in the recruiting process, since it was challenging to recruit flood mitigation expertise relative to diffuse pollution and biodiversity. This was probably due to overlaps in expertise, as stakeholders initially contacted for their primary expertise in flood mitigation, had even greater self-reported expertise in biodiversity or diffuse pollution. Academics, for instance, find their scientific niche not only through exceptional depth of knowledge of a specialized research domain, but also through their breadth of knowledge, with a topical scope spanning multiple knowledge domains (Bateman & Hess 2015). A potential weakness of eliciting expert opinion of the effectiveness of AEMs via BWS, particularly using an online platform, is that participants need to have some experience of the AEMs to be evaluated, and may therefore be biased towards favouring those schemes that they are more familiar with (Finn *et al.* 2009). The high degree of similarity between experts and non-experts highlighted here, however, suggests the experts were not significantly biased or pushing their own agenda linked to their field of interest. The only statistically significant difference was expert stakeholders scoring the ‘provide alternative drinking water sources for livestock’ AEM higher than non-expert stakeholders in the stakeholder group with expertise in diffuse pollution. This may be due to the diffuse pollution experts’ knowledge that, although livestock poaching near water courses only degrades a very small area within a catchment, it significantly contributes to bank erosion and nutrient enrichment of streams (Reaney *et al.* 2019).

'Riparian buffer zones' was the AEM that ranked highest for delivering cumulative benefits associated with reduced diffuse pollution, increased biodiversity and flood mitigation. Furthermore, riparian buffer zones provide a wide range of ecosystem services with benefits extending beyond those scored in this methodology, e.g. stabilising river banks, or mitigating climate change induced rising stream temperatures (Malcolm *et al.* 2008; Cole *et al.* 2020). Riparian margins clearly have an important role to play in increasing broader ecosystem service provisioning of catchments and have the potential to help land owners deliver diffuse pollution, biodiversity and flood mitigation benefits while also remaining agriculturally productive (McCracken *et al.* 2012). Even if establishing riparian buffer zones may not lead to direct improvements in ecological outcomes of a water course, it can be viewed as a 'no regrets' measure as it likely moves the system in the right direction (Harris & Heathwaite 2012). Future application of the BWS methodology could focus on riparian buffer zone AEMs, particularly including measures involving riparian tree planting as it was the most mentioned by participants in the open-ended part of the survey.

Using a 'model' Scottish catchment for the BWS surveys had the advantage of allowing participants to generalise their responses, however, by definition this meant that the specific details of catchment characteristics or prior land use history and management decisions were lacking (Cuttle *et al.* 2016). The methodology had an added benefit of being able to recruit land and water management stakeholders with expertise from various catchments across the country. Had the methodology been deployed in an exemplar catchment where diffuse pollution, flooding and biodiversity issues were understood, the approach may have been more straightforward for the experts to engage with. Focusing on a particular catchment, however, only delivers knowledge and information relevant to that catchment in a case study format, although clearly broad lessons can be learned. In contrast, the approach adopted in our study captures a more generalised set of stakeholder views that are applicable across Scottish catchments more broadly. In future research, our methodology could be applied to specific catchments, for example in a Scottish Environment Protection Diffuse pollution priority catchments, or in a Demonstration Test Catchment in England and Wales (McGonigle *et al.* 2014).

The cumulative ecosystem benefits (incorporating diffuse pollution, biodiversity and flood mitigation benefits of AEMs) fell broadly in line with the multiple benefits elicited from the ranking exercise. Due to the design of the study it was not possible to ascertain to what extent the latter was influenced by the judgements from the BWS exercises made beforehand. Fundamentally, repeating questions in surveys does not lead to independent measurements due to memory effects, where respondents recall their original answer and their memory acts as a source of bias for the following estimates (Schwarz *et al.* 2020). In future applications of this methodology it might be interesting to include a control group that only completes the ranking exercise, however, this may be a squandering of an already relatively small pool of experts and further increase difficulty for recruitment. These findings highlight that to avoid interference with the BWS survey it is vital that when repeating this survey, the ordering of the questions should remain the same and ensure the ranking question is asked after the BWS exercises to avoid memory effects biasing the most crucial part of the survey.

BWS surveys require participants to have a good understanding of all options they are being shown in order to produce the correct utility scores. If AEMs are more discipline-specific or less common there may be uncertainty or lack of clarity among participants regarding their specific effectiveness. This can lead to larger variances in an AEM's utility scores, as was found in 'rural sustainable drainage systems'. This AEM might be expected to receive greater utility scores due to to the multiple benefits of sediment and nutrient retention, flood attenuation and biodiversity (Ockenden *et al.* 2012). The variability recorded therefore suggests a lack of clarity of what the AEM would entail for experts less familiar with flood alleviation. This was reinforced through feedback from participants who indicated that they were not certain what types of measures this option would include (e.g. field wetlands, or sediment traps in ditches etc.), which highlights that specific mitigation approaches that fall under different AEM options can take varied forms that are likely to perform differently under different field situations. Conducting the survey online had the benefits of low costs, a standardised design and the freedom for the participants to respond in their own time; however, a face-to-face approach would have enabled greater opportunity for explanation and may have increased the motivation and sense of responsibility of experts (Knol *et al.* 2010). Conducting our study face-to-face would have allowed further explanation on the 'rural sustainable drainage systems' AEM, but it would have also required greater resource in terms of time and effort, which is a common trade-off when designing the collection of data via survey approaches (Nayak & Narayan 2019).

The comparative nature of selecting the best and the worst AEM is a strength of BWS, but it means the scores are somewhat arbitrary. In BWS surveys, respondents are not asked to indicate their views on the absolute importance of the items, as they are designed to determine a relative rank between the choices (Orme 2010). BWS studies may, however, include anchoring points as a solution to the relativity of utility scores (Chrzan & Orme 2019). This could be an AEM with specifically stated benefits, e.g. a sedimentation pond which removes a specified amount of sediment, provides habitat designed for particular biodiversity benefits and has capacity to hold a certain amount of flood water. There is also scope to extend the BWS survey to provide more specific details on the AEMs and to incorporate a greater number of ecosystem service benefits, such as benefits for climate change mitigation or for enhancing landscape features.

Capturing uncertainty in participants' responses could identify the AEMs that are not just the most likely to provide multiple benefits, but also AEMs with most certain positive impacts (Jones *et al.* 2018). This may be incorporated into our methodology by carrying out multiple BWS exercises, asking respondents how the AEMs would likely perform under best-case, most-likely and worst-case conditions. Scenario planning may also be included to elicitate perceived multiple benefits under uncertain future climate change and land use scenarios (Zhang *et al.* 2017). Bayesian belief networks are another tool that aid decision-making under uncertainty by capitalising on expert knowledge to fill gaps in socio-ecological modelling (Smith *et al.* 2018). The BWS methodology presented here may be used as a preliminary scoping exercise to select AEMs to include in Bayesian belief networks analysis. Hence the ability of the most promising AEMs to provide multiple ecosystem benefits could be further specified depending on different environmental and socio-economic variables, such as rainfall, topography, soil and land cover type (McVittie *et al.* 2015).

Implications for land and water management

Around the world, agri-environment schemes have often prioritised preserving biodiversity rather than other benefits, such as water quality or flood mitigation (Clements *et al.* 2021). European and North American countries are committing billions of dollars annually to AEM, but studies evaluating biodiversity conservation effectiveness of AEMs show mixed results, and fewer than 15% of these studies included any measure of cost-effectiveness (Ansell *et al.* 2016; Whittingham 2007). Besides understanding the environmental outcomes and the cost-effectiveness of AEMs, socio-cultural factors that might influence the quality of engagement with the schemes and the social well-being impact of AEMs also present a research gap (Mills *et al.* 2021). Farmer participation in AEMs may not only rely on payments offered vs. effort for adoption but also on the importance farmers give to other considerations, such as environmental effect or the production potential of land (van Herzele *et al.* 2013). Although not within the scope of this study, understanding the economic and cultural dimensions of AEMs will be vital for maximising ecosystem service benefits.

Expanding the knowledge base of the effectiveness of AEMs is important; however, there is also a need for creating schemes based on current best knowledge and recognising the value of expert knowledge for making assessments in data-poor environments (Elliott *et al.* 2007). Exploring how experts and non-experts rank AEMs delivers useful information for decision-making to support catchment management, but expertise is available from a range of sources. Local stakeholders spending the majority of their time in their local area will have a greater sense and appreciation of the landscape, in addition to their respective professional expertise (Lane *et al.* 2006). This knowledge should be made use of when implementing an AEM strategy. Farmers and land managers for instance, carry knowledge and experience from daily land management practices, going beyond what may be scientifically observed (Oliver *et al.* 2012). When implementing strategies for delivering multiple environmental benefits from AEMs it is therefore vital to include stakeholder knowledge. So, while the approach holds promise for prioritising AEM options to be promoted at the national-scale through programmes of measures, there are also opportunities for BWS to capture the views from local farmers and land managers. An example of smaller-scale applications of the methodology could be aiding decision-making for collaborative AEMs where multiple farm units implement measures to make them effective at the landscape scale (Mckenzie *et al.* 2013). The United Kingdom leaving the European Union presents a substantial opportunity to introduce AEM programmes with increased flexibility, local targeting, practicality and output-based approaches (Klaar *et al.* 2020). However, catchment management continues to be mostly driven by top-down decision-making, resulting in uneven involvement of different stakeholders in land and water management and poor maintenance of measures following implementation (Rollason *et al.* 2018). Incorporating effective participatory methodologies with the right stakeholders at appropriate scales will significantly impact the quality and success of any AEM programmes.

6.10. Conclusion

Agri-environment schemes were introduced in the United Kingdom in the 1980s as a response to the widespread environmental damage and species decline caused by post-war farming practices. Moving on from a predominate focus on biodiversity loss, schemes have been retrofitted to meet emerging issues of agricultural land management over the years, however, academic research has an opportunity to contribute towards the co-design of the most effective AEMs (Clements *et al.* 2021).

In this study we have used expert stakeholder-derived data to consider the effectiveness of specific agri-environment measures to decrease diffuse pollution to downstream water courses, increase habitat quality to support biodiversity and attenuate flood waters. There is an opportunity for innovative and engaging participatory approaches to play a key role in assessing AEMs effectiveness for multiple ecosystem service provisioning, as well as playing a part in including expert stakeholder knowledge in decision-making and providing a tool to foster mutual understanding. Multi-stakeholder participatory approaches can reveal hidden "win-win" solutions in landscape-scale AEM assessments which may otherwise be difficult or even impossible to determine.

7. Synthesis and Findings

Many of the most pressing challenges in water management are 'wicked problems' as they extend beyond traditional scales of analysis and management, and pose new uncertainties for decision-making (Cosens *et al.* 2014; Lintern *et al.* 2020). There is a need, therefore, to move beyond fragmented and single-issue driven responses to land and water management and towards holistic approaches and multi-level, integrated catchment management (ICM, Rouillard & Spray 2017). Key challenges for ICM are to establish how stakeholder knowledge can be used within existing frameworks of knowledge creation to inform decision-making, but also to develop new mechanisms for social learning and shared decision-making (Rollason *et al.* 2018).

This thesis builds on the growing contribution of stakeholder engagement and participatory research in the field of catchment management. The research outputs deliver new insight and help to underpin the scientific evidence base and best practice for eliciting stakeholder knowledge and expert opinion. This was achieved through a series of novel stakeholder engagement methodologies designed to assess ecosystem service provisioning and trade-offs within catchments, as well as potential synergies and conflict between stakeholder groups to better manage competing demands on catchment resources. Each data chapter provides a novel application of a tool or approach in the context of generating wider catchment scale insight from across multiple stakeholder groups. Results deliver engagement tools, scientific insight but also have implications for water management policy and practical decision-making. The findings address the research priorities for integrated catchment management identified in chapter 1, which were to (1) assess the trade-offs and synergies between catchment uses and find ways to optimise landscape scale ecosystem service provision in Scottish catchments; (2) quantify how stakeholder views differ between key groups of stakeholders and across different catchments with diverse and contrasting geomorphologies, land cover types, stakeholder communities and land and water management pressures; (3) develop methods that can reduce stakeholder conflict by facilitating cooperation and building shared mutual understanding and (4) determine the practical relevance of the participatory methodologies for land and water management planning and decision-making. Table 7.1 summarises the research findings that map onto these aims and direction for future research. The following sections discuss the scientific implications of the key findings and identify future research needs. The synthesis draws together the theoretical, practical, and methodological strengths of the methodologies used and evaluates their limitations.

7.1. *Assess the trade-offs and synergies between catchment uses and find ways to optimise landscape scale ecosystem service provision in Scottish catchments.*

Eliciting a range of perspectives from land and water management stakeholders in Chapters 2-6 has allowed assessment of differences in views among stakeholder groups and quantification of otherwise implicit expertise and understanding which could be utilised to better manage competing demands on catchment resources. Participatory research approaches can help facilitate effective knowledge exchange among stakeholders (Galafassi *et al.* 2017), increase the inclusiveness and social acceptability of management decisions, and ensure they are implemented more effectively (Oliver et al. 2017). Such approaches are therefore vital for integrating social demands, which are often not taken into account, and hence may avoid potential conflict of natural resource use and management (García-Nieto *et al.* 2013).

Although expert judgements are more liable to biases than other techniques due to tendencies such as overconfidence and anchoring (Mach *et al.* 2017), they may share otherwise unmeasurable insider knowledge and perspectives and can provide logical arguments to support their judgements (Singh *et al.* 2017). Making use of ‘thought experiments’ (such as that used in Chapters 2, 3 and 6) capitalised on stakeholder experience and acquired instinct to capture estimations which could not have been measured in the field. And expert stakeholder assessments of land and water management in specific study catchments (Chapters 4 and 5) can interpolate or extrapolate variables that may not be measured directly, such as providing time-integrated assessments, as opposed to momentary snapshots (Martin *et al.* 2012).

Table 7.1. Summary of findings of this project mapped against research priorities identified during project formulation and future research needs.

Research priorities	**Key Findings**	**Future research**
Assess the trade-offs and synergies between catchment uses and find ways to optimise landscape scale ecosystem service provision in Scottish catchments. (1)	**Chapter 3:** Investigating the production possibility frontier curve of the trade-off between two catchment uses gave insights into how best to manage the trade-off, e.g. not to push a system past its threshold levels. Participants from across a range of different stakeholder typologies mentioned that their work aims towards providing greater agricultural outputs without compromising ecological quality. **Chapter 4:** The catchment heat maps of perceived conflict, combining all the stakeholder's responses, showed that perception of conflict in all three catchments was often congruent among participants, with up to ten stakeholders identifying the same localised hotspots in their catchment. Many of the participants stated that often conflicts arise due to there being a clear trade-off between two reasonable interests which are conflicting, hence making a 'win-win' solution unlikely. **Chapter 5:** The stakeholder mapping methodology identified core and peripheral stakeholders in the three study catchments. **Chapter 6:** 'Riparian buffer zones' were perceived as offering the greatest overall benefits for diffuse pollution, biodiversity and flood attenuation.	Extending the trade-off analysis from two ecosystem services to a matrix of multiple ecosystem services in catchments, e.g. water, food and energy provisioning as well as regulating and cultural services. Longitudinal analysis to track temporal shifts in stakeholder and/or catchment citizen perceptions over time. Incorporating certainty weightings into expert elicitation methodologies, such as setting upper and lower bounds on parameters to highlight best- and worst-case scenarios, rather than a single outcome. Scenario planning or Bayesian belief networks may also integrate uncertainty of future environmental and socio-economic variables. Enhanced utilisation of expert opinion in environmental modelling.
Quantify how stakeholder views differ among key groups of stakeholders. (2a)	**Chapter 3:** Four stakeholder groups perceived the trade-off demands of other stakeholder groups differently; the largest difference in perspectives was identified between Environmental Regulators and Farm Advisors. A number of stakeholders did not agree that a trade-off between intensifying agricultural production and decreasing ecological quality of a river existed to begin with. Catchment Scientists' responses indicated a more precise and balanced insight into the socio-ecological system compared to other groups. **Chapter 4:** Pressures on farmland management were only identified by Farm Advisors and Catchment Scientists and pressures from tourism and recreation were only identified by Environmental Regulator Staff and Catchment Scientists. Catchment Scientists were the only stakeholder group to identify all of the categories of conflict. **Chapter 5:** Farm Advisors were up to three times as likely as participants from other stakeholder groups to perceive stakeholders as having a negative influence on catchment management. No Farm Advisors perceived Farmers as having a negative influence on catchment management, whereas other participants regularly scored farmers negatively.	Extend methodologies used in this thesis to quantify views from a wider range of stakeholder typologies, i.e., non-governmental organisations or the public. In turn, deploy these methodologies as tools for capturing views contributed at river basin management consultations and recognise their role in promoting more effective public participation. Incorporating the Delphi method to support consensus building among stakeholders and allow participants to review the results together with their stakeholder group and/or with the entire participant group.

Quantify how stakeholder views differ across different catchments. (2b)	**Chapter 4:** Results from the regression modelling linked perceived conflict to coastal, urban and grassland land use. In the agricultural catchment, conflict was more widely identified across the catchment, and the mean proportion of perceived conflict was greater than observed in the other study catchments. **Chapter 5:** The stakeholder mapping methodology was able to identify differences in underlying land and water management issues between the study catchments.	Applying methodologies used in this thesis to quantify stakeholder views to a wider range of river catchments across the UK, EU and globally.
Develop methods that can reduce stakeholder conflict by facilitating cooperation and building shared mutual understanding. (3)	**Chapter 3:** The methodology allows visualisation of otherwise implicit differences in how stakeholder groups understood social demands. **Chapter 4:** Increasing dialogue among stakeholder groups would help different parties to understand each other's viewpoints and lead to stakeholder empowerment and network building aiding future cooperation as opposed to conflict. **Chapter 5:** A notable difference in expected and elicited results was that neither the UK nor the EU Commission was mentioned by more than one stakeholder across all three catchments. **Chapter 6:** The best-worst scaling methodology was effective at assessing multiple ecosystem service provisioning of the agri-environment measures and presents a robust approach for including expert stakeholder knowledge in decision-making and a possible tool to foster mutual understanding.	Carry out a Citizen's Jury to debate trade-offs, synergies, and multiple ecosystem service provisioning in a study catchment to provide a platform to facilitate social learning. Greater collaboration within the research community, (i.e. between catchment and marine objectives or between environmental, economic and social sciences) across political and scientific lines, as well as across national borders. Integrate local stakeholders into the research process, from design to implementation, by co-producing research strategies.
Determine the practical relevance of the participatory methodologies for land and water management planning and decision-making. (4)	**Chapter 3:** The trade-off assessment may be utilised in catchment management programmes as part of a participatory approach to engage stakeholders. In doing so it could promote discussion of otherwise implicit decision-making and build shared mutual understanding to facilitate future cooperation. **Chapter 4:** The Scottish Government's target of expanding forestry in Scotland by 10 000 ha per year was highlighted as a policy that is likely to exacerbate conflicts and land use competition in all the study catchments. **Chapter 5:** The methodology could be utilised for national-scale scoping exercises (e.g. selecting post Brexit agri-environment schemes), or for eliciting expertise from local farmers and land managers (e.g. aiding decision-making for collaborative agri-environment measures implemented by multiple farms).	The participatory conflict mapping methodology could be used on a national scale to identify 'complex conflict priority catchments', similar to identifying 'diffuse pollution priority catchments', where catchments that are experiencing multiple and exacerbating conflicts could benefit from a more holistic management of landscapes. Development of decision-making frameworks which incorporate 'outcome-oriented objective settings' can incorporate objectives such as maximising multiple ecosystem service provision or improving EU WFD ecological status as well as include socio-economic constraints, such as time, political context, governance and cost-effectiveness to select effective management options.

Highlighting synergies and “win-win” opportunities is vital for maximising ecosystem service provisioning from a finite catchment area. Riparian buffer zones were perceived as offering the greatest overall benefits for diffuse pollution, biodiversity and flood attenuation (Chapter 6). Riparian margins clearly have an important role to play in increasing broader ecosystem service provisioning of catchments and have the potential to help land owners deliver diffuse pollution, biodiversity and flood mitigation benefits while also remaining agriculturally productive (McCracken *et al.* 2012). In the conflict mapping exercise, participants stated that often conflicts arise due to there being a clear trade-off between two reasonable interests which are conflicting, hence making a ‘win-win’ solution unlikely. When facing land and water management issues with possibilities for conflict and no clear synergies or ‘win-win’- solutions, future research could incorporate the Delphi method into the methodologies presented here to support consensus building among stakeholders. It is an iterative structure that allows stakeholders to express their opinions in multiple rounds, where they may review their answers in light of other participants’ responses. This allows stakeholders to interact remotely and at different times to allow convergence of ideas and consensus building without biases that focus groups may cause, such as certain individuals and themes dominating the discussion (Giuffrida *et al.* 2019). Delphi approaches have for instance shown to be effective at collectively refining and vetting solutions and priorities to inform policy dialogue on addressing water quality issues in the context of climate change (Coleman *et al.* 2017).

The production possibility frontier methodology gave valuable insights on how the different stakeholder groups were prioritising the management of a land and water management trade-off (Chapter 3). Its design was limited to two catchment uses however. Extending the trade-off analysis to a matrix of multiple ecosystem services in catchments, e.g. water, food and energy provisioning as well as regulating and cultural services, could also present an exciting avenue for future research. Chapter 3 also allowed stakeholders to incorporate small, medium or large uncertainty intervals to their responses. There is a gap for more ICM tools that incorporate uncertainty estimation into expert elicitation methodologies. Certainty weightings, such as setting upper and lower bounds on parameters to highlight best- and worst-case scenarios, rather than a single outcome (Ahmed *et al.* 2018). Scenario planning or Bayesian belief networks also present effective methodologies for integrating uncertainty and expert views of future environmental and socio-economic variables (Shi *et al.* 2020). Finally, longitudinal analysis to track temporal shifts in the perceptions of a cohort of stakeholders over time, although clearly challenging to secure funding for, would represent frontier interdisciplinary research and better our understanding of the socio-ecological complexity of catchment systems.

There are also opportunities for greater integration within the research community, such as between catchment and marine objectives or between environmental, economic and social sciences and even IT and engineering to make participatory methodologies even more engaging. A particularly novel and interdisciplinary proposal is the ‘gamification’ of participatory tools, made possible by the design of gaming interfaces as a user-friendly front-end, i.e., a graphic user interface. This would allow a large number of participants to ‘play’ and explore the underpinning science associated with the complexity of multiple impacts of decision-making (Turner *et al.* 2016). Similarly, effective use of visual tools has potential to promote inclusive public participation (Roque de Oliveira & Partidário 2020). Interactive and immersive tabletop interfaces have the potential to enable stakeholders to interact with complex datasets in a natural way and promote collaborative interaction (Ens *et al.* 2021; Hughes *et al.* 2011).

7.2. *Quantify how stakeholder views differ among key groups of stakeholders and across different catchments with diverse and contrasting geomorphologies, land cover types, stakeholder communities and land and water management pressures.*

The most conspicuous differences among stakeholders often included the Farm Advisors group. They held opposing views on perceived trade-off demands of their own and other stakeholder groups, and some Farm Advisor participants did not agree that a trade-off between intensifying agricultural production and decreasing ecological quality of a river existed (Chapter 3). They were also up to three times as likely as participants from other stakeholder groups to perceive stakeholders as having a negative influence on catchment management; while no Farm Advisors perceived Farmers as having a negative influence on catchment management, whereas other participants regularly scored farmers negatively. Some studies report on a defensive attitude of farmers in regards to diffuse pollution and mention a lack of information on the reality and magnitude of the problems it causes (Martínez-Dalmau et al. 2021). As society's expectations of the farming community changes from a focus on food security, animal welfare, and landscape heritage to inclusion of broader social and environmental goals (Saunders 2016), however, farmers may become resistant to challenges to their identity of what makes a 'good farmer' (Thomas *et al.* 2019; Collins 2018). As point sources of pollution become more well-managed, agriculture is becoming the single largest contributor to diffuse pollution in Scotland (Waylen *et al.* 2015). Although the ecological downstream consequences of farm activities need to be understood and managed, there is a risk that stereotyping, distrust and scapegoating may cause conflict and impede the ability of stakeholders to collaborate effectively to solve land and water management issues (Curșeu & Schruijer 2017). Findings from chapter 3 identified the gulf between perceived land and water management attitudes being greatest between Environmental Regulators and Farm Advisors, two stakeholder groups whose close partnership will be essential to solving future land and water management issues. Adapting the production possibility trade-off methodology to the context of a focus group discussion between Environmental Regulators and Farm Advisors could present an opportunity for these key stakeholders to articulate their views in a non-confrontational and abstract setting as well as reflect on how accurately the data represents the realities in their catchments (Cebrián-Piqueras *et al.* 2017). This would promote discussion of otherwise implicit attitudes, and may help to build shared mutual understanding and facilitate future cooperation (Brunet *et al.* 2018). Catchment Scientists' responses indicated precise and balanced insights into the socio-ecological system compared to other stakeholders (Chapters 3, 4 & 5), which may reflect their role as outside observers, seeking unbiased, objective descriptions of reality (Rose & Parsons 2015). These findings indicate the special role researchers can play as brokers to promote stakeholder cooperation and knowledge sharing and aid implementation of innovative land management practice (Schröter *et al.* 2015). Farm Advisors also have a central role to play to act as brokers between the farming and the research communities as they are familiar with local land management and can speak the agricultural language is key to effective communication and knowledge exchange with the farmers (Skaalsveen et al. 2020; Mohamad Ibrahim et al. 2019). Arguably, Scientists and Farm Advisors focus on two complementary 'broker practices', managing the interactions between diverse knowledge systems. But whereas research has tools for dealing with uncertainty around the link between practice and outcome, Advisors have tools for dealing with uncertainty around multiple goals, highlighting the importance of their close collaboration in land and water management (Love *et al.* 2006).

Due to the survey design of chapters 3-5 and to be able to compare groups using statistics, at least three participants were needed within each catchment and stakeholder group, exacerbating the already inherently difficult recruitment of stakeholder participation in environmental management (Holifield & Williams 2019). This complication, however, proved worthwhile as the data collected in this series of studies provided unprecedented findings into how stakeholder views vary between different catchments and between key land and water management stakeholders. As these studies acted as a proof of concept, future applications of the methodologies may extend recruitment to a wider range of stakeholder typologies such as non-governmental organisations or the public. Both expert and non-expert stakeholders can improve results in assessing ecosystem services (Asah & Blahna 2020), but importantly opening participation to a diverse number of stakeholders can have their voice heard, as the environment matters to local people, both experts and non-experts (Klain *et al.* 2014).

7.3. *Develop methods that can reduce stakeholder conflict by facilitating cooperation and building shared mutual understanding.*

The engagement methodologies employed in chapters 2-6 involved the elicitation of expert data from individuals, using a variety of elicitation media. Individual interviews allowed more targeted questioning, explanation and feedback. Although group discussions can make disciplinary biases more explicit and discount redundant information through sharing of knowledge, they can become dominated by individuals and over-emphasise consensus (Krueger *et al.* 2012). We carried out an online survey, with the benefits of lower cost, standardisation and freedom for the participants to respond in their own time (Page *et al.* 2012); as well as face-to-face interviews, which leave more room for explanation and may increase the motivation and sense of responsibility of experts (Knol *et al.* 2010). Although our methodologies focused on quantifying and characterising expert stakeholder views, stakeholder relationships, conflict and likely solutions, additional qualitative data was useful to cross-validate the quantitative results and to support wider understanding of the stakeholder's views. Stakeholders may also welcome a mixed approach as it indicates a desire to fully understand their perspective (Dick *et al.* 2018). The methodologies allowed visualisation of otherwise implicit differences in how stakeholder groups understood social demands, conflict and synergies in catchment management. Using maps to engage stakeholders has obvious benefits for eliciting location-based information and may also be a more accessible medium for experts with a strong connection to the landscape such as farmers (Oliver *et al.* 2012). Other elicitation media, such as petal diagrams, production possibility frontier trade-off graphs and the best-worst scaling survey presented participants with novel exercises to engage with; however, it is vital that the results from the mixed methods research presented here are used to inform rather than end a discussion among stakeholders (Verhagen *et al.* 2018). An effective way of facilitating such a discussion could be a Citizen's Jury to debate trade-offs, synergies, and multiple ecosystem service provisioning (Fish *et al.* 2013). Here, results from chapters 2-6 and other findings could be used as evidence to present to local citizens by expert stakeholders to provide a platform to facilitate social learning around the challenges of catchment management and competing demands on catchment resources.

Social distancing measures imposed to stop the spread of Covid-19 have severely affected the process of stakeholder participation. Home-working and the lack of 'water cooler talk' or meeting each other at events has severely restricted opportunities for researchers and stakeholders to collaborate face-to-face. Digital engagement has become the new norm and can offer a greater continuity of exchange at reduced effort and with a much lower carbon footprint (Köpsel et al. 2021). Although there might be an opportunity to engage online with a wider group of stakeholders, there is a risk of excluding certain societal groups and more local stakeholders.

Knowledge co-production with stakeholders has now become a common practice, or at least a common ideal, to look for solutions to water governance as well as for water-related research (Brugnach & Özerol 2019). This was accelerated by requirements the Water Framework Directive made on stakeholder participation; however, there is a need to find appropriate ways of stakeholder involvement that make effective use of stakeholder knowledge and allow for interactive communication (Llopis-Albert *et al.* 2017; Rimmert *et al.* 2020). Stakeholders views may also be integrated into the research process at earlier stages, from design to implementation, by co-producing research strategies (Singletary & Sterle 2020). Implementation of collaboratively determined research questions are made extremely difficult due to the structure of much academic funding, where there is often a need for finely specified research questions in proposals (Bell & Pahl 2018). Hence, a different approach to research funding is needed in order to support the complex partnerships necessary for co-production (Redman *et al.* 2021). Collaborative research requires experts skilled in interpersonal communication and group procedure, and the sustained commitment of the active participation of researchers, stakeholders, and their respective institutions (Gober 2018). Stakeholders in particular may experience fatigue if they are asked repeatedly to volunteer substantial time from their daily management responsibilities to participate in collaborative research projects (Lemos *et al.* 2018). It is therefore vital to introduce processes to improve collaborative efficacy when designing and decision making in multi-stakeholder co-creation (Jones 2018).

7.4. *Determine the practical relevance of the participatory methodologies for land and water management planning and decision-making*.

The findings from this thesis, identifying underlying stakeholder connections, implicit understandings and misunderstandings within Scottish catchments signify the importance for research into stakeholder views of land and water management, and highlight the necessity of participatory approaches for effective catchment management. Successful land and water management utilising integrated and interdisciplinary approaches involves identifying and developing a range of a diverse suites of technical, design, policy and social measures (Rogers *et al.* 2020) and requires improved communication and extended engagement between governments and stakeholders to better link strategic decision-making and operational practice (Rouillard & Spray 2017). The finding from Chapter 4, for instance, that the Scottish Government target of expanding forestry in Scotland by 10000 ha per year was highlighted as a policy that is likely to exacerbate conflicts and land use competition in the study catchments, exemplifies the risk for conflict of policy measures, despite woodland expansion being generally considered by stakeholders to be an overall positive initiative in Scotland (Burton *et al.* 2018). Given the devolution of responsibilities for different management bodies in the UK such as for flood protection, biodiversity conservation and river basin management planning it may difficult to encourage managers beyond the remit of their responsibilities (Rust & Venn 2018). To maximise ecosystem functioning of catchments it will be vital, however, to abandon a siloed approach to water management and integrate decision-making, not just within governance structures, but also across political and scientific lines, as well as across national borders and broader stakeholder communities. To encourage cross-governance collaboration, steering authorities could lower implementation barriers by setting incentives, such as funding programmes, that offer additional advantages for both collaborating parties (Schröder *et al.* 2020). Scotland's relatively joined-up governance structure has already enabled policymakers and stakeholders to work together effectively to build trust and cooperation, for example to facilitate the adoption of stricter measures for tackling diffuse pollution which was not achieved in England (De Vito *et al.* 2020). Arguably, this favourable governance structure, it being a relatively manageably sized country, and the existing focus of Scotland as a 'Hydro Nation' puts Scotland in an exceptional position to become a pioneer for ICM. Fostering multi-stakeholder platforms in Scottish catchments may strengthen existing catchment management groups, create new networks for stakeholder cooperation and negotiation and allow further integration of land and water management (Warner 2007).

Figure 7.1 illustrates the complexity of interactions within socio-ecological systems and how stakeholder engagement methods can help to quantify stakeholder views to aid understanding of the social and ecological system in a river catchment. Due to the inherent complexities of catchment systems, integrated catchment management may not be implemented in a standardised way but should start with careful analysis of the local context, and existing governance arrangements and governmentalities (Watson *et al.* 2019). The conflict mapping and stakeholder network mapping technique from chapters 4 and 5 present two engaging methodologies that may be utilised for evaluating stakeholder associations and the spatial context of land and water management issues. The participatory conflict mapping methodology could also be used on a national scale to identify 'complex conflict priority catchments', similar to identifying 'diffuse pollution priority catchments', where catchments that are experiencing multiple and exacerbating conflicts could benefit from a more holistic management of landscapes. The trade-off assessment may be utilised in catchment management programmes to help resolve issues at local scales through to catchment scales and engage multiple stakeholders. In doing so it could promote discussion of otherwise implicit decision-making and build shared mutual understanding to facilitate future cooperation. The BWS methodology could be utilised for national-scale scoping exercises, such as for selecting post Brexit agri-environment schemes; or for eliciting expertise from local farmers and land managers, for example to aid decision-making for collaborative agri-environment measures implemented by multiple farms (Mckenzie *et al.* 2013). Co-producing agri-environment schemes with land and water

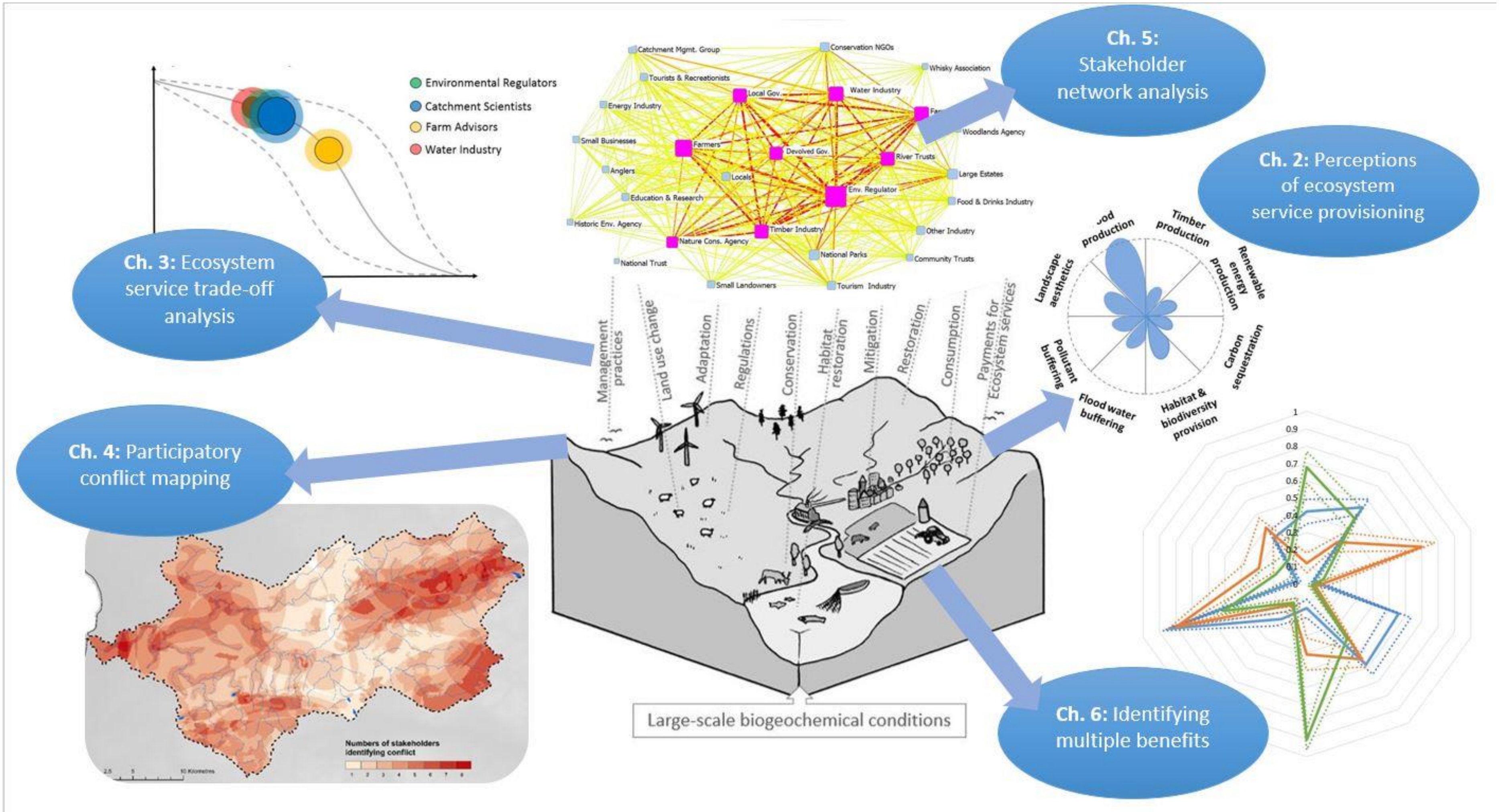


Figure 7.1: Examples of results from the stakeholder engagement methodologies developed for this thesis aimed to aid understanding of the social and ecological system in a river catchment, assess trade-offs and synergies between catchment uses, find ways to optimise landscape scale ecosystem service provision, and develop methods that can reduce stakeholder conflict by facilitating cooperation and building shared mutual understanding.

management stakeholders may also be an opportunity to allow farmers to embed their understanding of landscape stewardship and their landscape values into the effort of managing agriculture for multiple ecosystem service provisioning (Raymond *et al.* 2016).

Due to the complexities of socio-ecological systems, uncertainties around the magnitude of catchment management issues and how best to mitigate them will remain, however strategically placed nature-based measures for multiple ecosystem service provisioning are likely the best option to achieve holistic water resource management (Hewett *et al.* 2009). River restoration projects have shown to be able to increase the ability of catchments to provide cultural and regulating ecosystem services, without significantly affecting provisioning services (Vermaat *et al.* 2016) and may even have substantial direct economic benefits and non-market values (Gerner *et al.* 2018). To make use of such "win-win" scenarios, there is a need to translate interdisciplinary research on catchments into informed decision-support tools to allow policy makers, communities, and individual stakeholders to make better informed decisions. Including these in existing structures, such as the Water Framework Directive River Basin Management Cycles, or agri-environment funding cycles, and building upon positive pre-existing relationships, such as catchment management groups, may be the best approach to ensure effective stakeholder participation and develop strong partnerships among stakeholders (Barnhart *et al.* 2018). Development of decision-making frameworks which incorporate 'outcome-oriented objective settings' can incorporate objectives such as maximising multiple ecosystem service provision or improving EU Water Framework Directive ecological status as well as include socio-economic constraints, such as time, political context, governance and cost-effectiveness to select effective management options.

7.5. Conclusion

Integrated catchment management necessarily requires land and water to be managed in a holistic manner, that identifies pressures on the water environment, recognises potential for conflict between the interests of users and allows stakeholders to work together to agree common objectives and implement solutions. Findings from this thesis highlight the importance on co-producing knowledge in catchment research, add significantly to the evidence base of transdisciplinary catchment research and supply a range of novel and engaging methodologies. Although this thesis focused on Scottish catchments, there is great potential for international transferability as both the findings and approaches have widespread applicability and relevance to the socio-ecological functioning of catchments worldwide. The methods developed here have potential to be used by academics as preliminary stages within their research, or as direction for novel research discussed throughout this chapter. The approaches may also be utilised by catchment management groups and could aid land and water managers and decision-makers consider complex stakeholder relationships and ecosystem service interactions into the process of ICM.

References

Abram, N. K., Meijaard, E., Wilson, K. A., Davis, J. T., Wells, J. A., Ancrenaz, M., Budiharta, S., Durrant, A., Fakhruzzi, A., Runting, R. K., Gaveau, D., and Mengersen, K. (2017). Oil palm–community conflict mapping in Indonesia: A case for better community liaison in planning for development initiatives. *Applied Geography*, **78**, 33–44.

Adams, E. A., Kuusaana, E. D., Ahmed, A., and Campion, B. B. (2019). Land dispossessions and water appropriations: Political ecology of land andwater grabs in Ghana. *Land Use Policy*, **87**(104068).

Adams, V. M., Álvarez-Romero, J. G., Carwardine, J., Cattarino, L., Hermoso, V., Kennard, M. J., Linke, S., Pressey, R. L., and Stoeckl, N. (2014). Planning across freshwater and terrestrial realms: Cobenefits and tradeoffs between conservation actions. *Conservation Letters*, pp. 425–440.

Ahmadi, A., Kerachian, R., Rahimi, R., and Emami Skardi, M. J. (2019). Comparing and combining Social Network Analysis and Stakeholder Analysis for natural resource governance. *Environmental Development*, **32**. doi:10.1016/j.envdev.2019.07.001

Ahmed, A., Woulds, C., Drake, F., and Nawaz, R. (2018). Beyond the tradition: Using Fuzzy Cognitive Maps to elicit expert views on coastal susceptibility to erosion in Bangladesh. *Catena*, **170**. doi:10.1016/j.catena.2018.06.003

Aitken, M. N. (2003). Impact of agricultural practices and river catchment characteristics on river and bathing water quality. In *Water Science and Technology*, Vol. 48, pp. 217–224.

Ajiero, I., and Campbell, D. (2018). Benchmarking Water Use in the UK Food and Drink Sector: Case Study of Three Water-Intensive Dairy Products. *Water Conservation Science and Engineering*, **3**(1). doi:10.1007/s41101-017-0036-0

Albert, J. S., Destouni, G., Duke-Sylvester, S. M., Magurran, A. E., Oberdorff, T., Reis, R. E., Winemiller, K. O., and Ripple, W. J. (2021). Scientists' warning to humanity on the freshwater biodiversity crisis. *Ambio*, **50**(1), 85–94.

Alidoosti, Z., Ahmad sadegheih, Govindan, K., Pishvaee, M. S., Mostafaeipour, A., and Hossain, A. K. (2021). Social sustainability of treatment technologies for bioenergy generation from the municipal solid waste using best worst method. *Journal of Cleaner Production*, **288**. doi:10.1016/j.jclepro.2020.125592

Álvarez-Romero, J. G., Adams, V. M., Pressey, R. L., Douglas, M., Dale, A. P., Auge, A. A., Ball, D., Childs, J., Digby, M., Dobbs, R., Gobius, N., Hinchley, D., Lancaster, I., Maughan, M., and Perdrisat, I. (2015). Integrated cross-realm planning: A decision-makers' perspective. *Biological Conservation*, Elsevier B.V., pp. 799–808.

Andersson, E., Nykvist, B., Malinga, R., Jaramillo, F., and Lindborg, R. (2015). A social–ecological analysis of ecosystem services in two different farming systems. *Ambio*, **44**(1), 102–112.

Ansell, D., Freudenberger, D., Munro, N., and Gibbons, P. (2016). The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review. *Agriculture, Ecosystems and Environment*. doi:10.1016/j.agee.2016.04.008

Aruga, K., Bolt, T., and Pest, P. (2021). Energy policy trade-offs in Poland: A best-worst scaling discrete choice experiment. *Energy Policy*, **156**. doi:10.1016/j.enpol.2021.112465

Asah, S. T., and Blahna, D. J. (2020). Involving Stakeholders' Knowledge in Co-designing Social Valuations of Biodiversity and Ecosystem Services: Implications for Decision-Making. *Ecosystems*, **23**(2). doi:10.1007/s10021-019-00405-6

Asquith, N. M., Vargas, M. T., and Wunder, S. (2008). Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological Economics*, **65**(4), 675–684.

Austin, Z., McVittie, A., McCracken, D., Moxey, A., Moran, D., and White, P. C. L. (2016). The co-benefits of biodiversity conservation programmes on wider ecosystem services. *Ecosystem Services*, **20**, 37–43.

Austrheim, G., Speed, J. D. M., Evju, M., Hester, A., Holand, Ø., Loe, L. E., Martinsen, V., Mobæk, R., Mulder, J., Steen, H., Thompson, D. B. A., and Mysterud, A. (2016). Synergies and trade-offs between ecosystem services in an alpine ecosystem grazed by sheep – An experimental approach. *Basic and Applied Ecology*, **17**(7), 596–608.

Barnaud, C., Corbera, E., Muradian, R., Salliou, N., Sirami, C., Vialatte, A., Choisis, J. P., Dendoncker, N., Mathevet, R., Moreau, C., Reyes-García, V., Boada, M., Deconchat, M., Cibien, C., Garnier, S., Maneja, R., and Antona, M. (2018). Ecosystem services, social interdependencies, and collective action: A conceptual framework. *Ecology and Society*, **23**(1). doi:10.5751/ES-09848-230115

Barnes, A. P., Willock, J., Hall, C., and Toma, L. (2009). Farmer perspectives and practices regarding water pollution control programmes in Scotland. *Agricultural Water Management*, **96**(12), 1715–1722.

Barnes, M. L., Bodin, Ö., Guerrero, A. M., McAllister, R. R. J., Alexander, S. M., and Robins, G. (2017). The social structural foundations of adaptation and transformation in social–ecological systems. *Ecology and Society*, **22**(4). doi:10.5751/ES-09769-220416

Barnhart, B. L., Golden, H. E., Kasprzyk, J. R., Pauer, J. J., Jones, C. E., Sawicz, K. A., Hoghooghi, N., Simon, M., McKane, R. B., Mayer, P. M., Piscopo, A. N., Ficklin, D. L., Halama, J. J., Pettus, P. B., and Rashleigh, B. (2018). Embedding co-production and addressing uncertainty in watershed modeling decision-support tools: Successes and challenges. *Environmental Modelling and Software*. doi:10.1016/j.envsoft.2018.08.025

Batáry, P., Dicks, L. V., Kleijn, D., and Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, **29**(4), 1006–1016.

Bateman, I. J., Harwood, A. R., Mace, G. M., Watson, R. T., Abson, D. J., Andrews, B., Binner, A., Crowe, A., Day, B. H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A., … Termansen, M. (2013). Bringing ecosystem services into economic decision-making: land use in the United Kingdom. *Science*, **341**(6141), 45–50.

Bateman, T. S., and Hess, A. M. (2015). Different personal propensities among scientists relate to deeper vs. broader knowledge contributions. *Proceedings of the National Academy of Sciences of the United States of America*, **112**(12). doi:10.1073/pnas.1421286112

Bell, D. M., and Pahl, K. (2018). Co-production: towards a utopian approach. *International Journal of Social Research Methodology*, **21**(1). doi:10.1080/13645579.2017.1348581

Bellver-Domingo, A. Hernández-Sancho, F. Molinos-Senante, M. (2016). A review of Payment for Ecosystem Services for the economic internalization of environmental externalities: A water perspective. *Geoforum*, **60**, 115–118.

Bennett, E. M., Peterson, G. D., and Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters*, **12**, 1–11.

Berger, T., Birner, R., Mccarthy, N., Díaz, J., and Wittmer, H. (2007). Capturing the complexity of water uses and water users within a multi - Agent framework. In *Water Resources*

Management, Vol. 21, pp. 129–148.

Bernhardt, E. S., and Palmer, M. A. (2011). Evaluating river restoration. *Ecological Applications*, p. 1925.

Biggs, E. M., Bruce, E., Boruff, B., Duncan, J. M. A., Horsley, J., Pauli, N., McNeill, K., Neef, A., Van Ogtrop, F., Curnow, J., Haworth, B., Duce, S., and Imanari, Y. (2015). Sustainable development and the water-energy-food nexus: A perspective on livelihoods. *Environmental Science and Policy*, **54**, 389–397.

Bivand, R., Keitt, T., and Rowlingson, B. (n.d.). rgdal: Bindings for the Geospatial Data Abstraction Library. R package version 1.1-8.

Bodin, Ö. (2017). Collaborative environmental governance: Achieving collective action in social-ecological systems. *Science*. doi:10.1126/science.aan1114

Boelens, R. (2014). Cultural politics and the hydrosocial cycle: Water, power and identity in the Andean highlands. *Geoforum*, **57**, 234–247.

Bohnet, I. C., Roebeling, P. C., Williams, K. J., Holzworth, D., van Grieken, M. E., Pert, P. L., Kroon, F. J., Westcott, D. A., and Brodie, J. (2011). Landscapes Toolkit: An integrated modelling framework to assist stakeholders in exploring options for sustainable landscape development. *Landscape Ecology*, **26**(8), 1179–1198.

Borowski, I., and Hare, M. (2007). Exploring the gap between water managers and researchers: Difficulties of model-based tools to support practical water management. *Water Resources Management*, **21**(7), 1049–1074.

Bouleau, G., and Pont, D. (2015). Did you say reference conditions? Ecological and socio-economic perspectives on the European Water Framework Directive. *Environmental Science and Policy*, **47**, 32–41.

Brown, G., and Fagerholm, N. (2014). Empirical PPGIS/PGIS mapping of ecosystem services: A review and evaluation. *Ecosystem Services*, **13**, 119–133.

Brown, G., and Raymond, C. M. (2014). Methods for identifying land use conflict potential using participatory mapping. *Landscape and Urban Planning*, **122**, 196–208.

Brown, G., Reed, P., and Raymond, C. M. (2020). Mapping place values: 10 lessons from two decades of public participation GIS empirical research. *Applied Geography*. doi:10.1016/j.apgeog.2020.102156

Brown, G., Strickland-Munro, J., Kobryn, H., and Moore, S. A. (2017). Mixed methods participatory GIS: An evaluation of the validity of qualitative and quantitative mapping methods. *Applied Geography*, **79**, 153–166.

Brugnach, M., and Özerol, G. (2019). Knowledge co-production and transdisciplinarity: Opening Pandora's box. *Water (Switzerland)*. doi:10.3390/w11101997

Brunet, L., Tuomisaari, J., Lavorel, S., Crouzat, E., Bierry, A., Peltola, T., and Arpin, I. (2018). Actionable knowledge for land use planning: Making ecosystem services operational. *Land Use Policy*, **72**, 27–34.

Bryan, B. A., Raymond, C. M., Crossman, N. D., and King, D. (2011). Comparing spatially explicit ecological and social values for natural areas to identify effective conservation strategies. *Conservation Biology : The Journal of the Society for Conservation Biology*, **25**(1), 172–81.

Bryan, B. A., Raymond, C. M., Crossman, N. D., and Macdonald, D. H. (2010). Targeting the management of ecosystem services based on social values: Where, what, and how? *Landscape and Urban Planning*, **97**(2), 111–122.

Burton, V., Metzger, M. J., Brown, C., and Moseley, D. (2018). Green Gold to Wild Woodlands; understanding stakeholder visions for woodland expansion in Scotland. *Landscape Ecology*, 1–21.

Butler, J. R. a., Wong, G. Y., Metcalfe, D. J., Honzák, M., Pert, P. L., Rao, N., van Grieken, M. E., Lawson, T., Bruce, C., Kroon, F. J., and Brodie, J. E. (2013). An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef, Australia. *Agriculture, Ecosystems & Environment*, **180**, 176–191.

Canedoli, C., Bullock, C., Collier, M. J., Joyce, D., and Padoa-Schioppa, E. (2017). public participatory mapping of cultural ecosystem services: Citizen perception and park management in the Parco Nord of Milan (Italy). *Sustainability (Switzerland)*, **9**(6). doi:10.3390/su9060891

Carvalho, L., Mcdonald, C., de Hoyos, C., Mischke, U., Phillips, G., Borics, G., Poikane, S., Skjelbred, B., Solheim, A. L., Van Wichelen, J., and Cardoso, A. C. (2013). Sustaining recreational quality of European lakes: Minimizing the health risks from algal blooms through phosphorus control. *Journal of Applied Ecology*, **50**(2), 315–323.

Castro, A. J., Verburg, P. H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaughn, C. C., and López, E. (2014). Ecosystem service trade-offs from supply to social demand: A landscape-scale spatial analysis. *Landscape and Urban Planning*, **132**, 102–110.

Cavender-Bares, J., Polasky, S., King, E., and Balvanera, P. (2015). A sustainability framework for assessing trade-offs in ecosystem services. *Ecology and Society*, **20**(1), 17.

Cebrián-Piqueras, M. A., Karrasch, L., and Kleyer, M. (2017). Coupling stakeholder assessments of ecosystem services with biophysical ecosystem properties reveals importance of social contexts. *Ecosystem Services*, **23**, 108–115.

Chrzan, K., and Orme, B. K. (2019). *Applied MaxDiff: A practitioner's guide to best-worst scaling.*, Sawtooth Software.

Cinner, J. E., and Barnes, M. L. (2019). Social Dimensions of Resilience in Social-Ecological Systems. *One Earth*. doi:10.1016/j.oneear.2019.08.003

Clements, J., Lobley, M., Osborne, J., and Wills, J. (2021). How can academic research on UK agri-environment schemes pivot to meet the addition of climate mitigation aims? *Land Use Policy*, **106**. doi:10.1016/j.landusepol.2021.105441

Cole, L. J., Stockan, J., and Helliwell, R. (2020). Managing riparian buffer strips to optimise ecosystem services: A review. *Agriculture, Ecosystems and Environment*. doi:10.1016/j.agee.2020.106891

Coleman, S., Hurley, S., Koliba, C., and Zia, A. (2017). Crowdsourced Delphis: Designing solutions to complex environmental problems with broad stakeholder participation. *Global Environmental Change*, **45**. doi:10.1016/j.gloenvcha.2017.05.005

Collins, H. M. (2018). *Dairy farmers' responses to water quality interventions: a case study in the Manawatu-Wanganui region of New Zealand*, Massey University.

Coppes, J., Ehrlacher, J., Thiel, D., Suchant, R., and Braunisch, V. (2017). Outdoor recreation causes effective habitat reduction in capercaillie Tetrao urogallus: a major threat for geographically restricted populations. *Journal of Avian Biology*, **48**(12), 1583–1594.

Cord, A. F., Bartkowski, B., Beckmann, M., Dittrich, A., Hermans-Neumann, K., Kaim, A., Lienhoop, N., Locher-Krause, K., Priess, J., Schröter-Schlaack, C., Schwarz, N., Seppelt, R., Strauch, M., Václavík, T., and Volk, M. (2017). Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. *Ecosystem Services*, pp. 264–272.

Cordingley, J. E., Newton, A. C., Rose, R. J., Clarke, R. T., and Bullock, J. M. (2016). Can landscape-scale approaches to conservation management resolve biodiversity-ecosystem service trade-offs? *Journal of Applied Ecology*, **53**(1), 96–105.

Cosens, B., Gunderson, L., and Chaffin, B. (2014). the Adaptive Water Governance Project: Assessing Law, Resilience and Governance in Regional Socio-Ecological Water Systems Facing a Changing Climate. *Idaho Law Review*, **51**(1).

Costa, M. H., Botta, A., and Cardille, J. a. (2003). Effects of large-scale changes in land cover on the discharge of the Tocantins River, Southeastern Amazonia. *Journal of Hydrology*, **283**(1–4), 206–217.

Costanza, R., Arge, R., Groot, R. De, Farberk, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., Neill, R. V. O., Paruelo, J., Raskin, R. G., and Suttonkk, P. (1997). The value of the world ' s ecosystem services and natural capital. *Nature*, **387**(May), 253–260.

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., and Grasso, M. (2017). Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosystem Services*, pp. 1–16.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., and Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, **26**, 152–158.

Couto, T. B. A., and Olden, J. D. (2018). Global proliferation of small hydropower plants – science and policy. *Frontiers in Ecology and the Environment*, pp. 91–100.

Cronkleton, P., Albornoz, M. A., Barnes, G., Evans, K., and de Jong, W. (2010). Social geomatics: Participatory forest mapping to mediate resource conflict in the Bolivian Amazon. *Human Ecology*, **38**(1), 65–76.

Cross, P., Rigby, D., and Edwards-Jones, G. (2012). Eliciting expert opinion on the effectiveness and practicality of interventions in the farm and rural environment to reduce human exposure to Escherichia coli O157. *Epidemiology and Infection*, **140**(4). doi:10.1017/S0950268811001257

Cumming, G. S., Buerkert, A., Hoffmann, E. M., Schlecht, E., Von Cramon-Taubadel, S., and Tscharntke, T. (2014). Implications of agricultural transitions and urbanization for ecosystem services. *Nature*, pp. 50–57.

Curșeu, P. L., and Schruijer, S. G. (2017). Stakeholder diversity and the comprehensiveness of sustainability decisions: the role of collaboration and conflict. *Current Opinion in Environmental Sustainability*. doi:10.1016/j.cosust.2017.09.007

Cusack, J. J., Bradfer-Lawrence, T., Baynham-Herd, Z., Castelló y Tickell, S., Duporge, I., Hegre, H., Moreno Zárate, L., Naude, V., Nijhawan, S., Wilson, J., Zambrano Cortes, D. G., and Bunnefeld, N. (2021). Measuring the intensity of conflicts in conservation. *Conservation Letters*. doi:10.1111/conl.12783

Cuttle, S. P., Newell-Price, J. P., Harris, D., Chadwick, D. R., Shepherd, M. A., Anthony, S. G. A., Macleod, C. J. A., Haygarth, P. M., and Chambers, B. J. (2016). A method-centric “User Manual” for the mitigation of diffuse water pollution from agriculture. *Soil Use and Management*, **32**. doi:10.1111/sum.12242

D'Agostino, D., Borg, M., Hallett, S. H., Sakrabani, R. S., Thompson, A., Papadimitriou, L., and Knox, J. W. (2020). Multi-stakeholder analysis to improve agricultural water management policy and practice in Malta. *Agricultural Water Management*. doi:10.1016/j.agwat.2019.105920

Darvill, R., and Lindo, Z. (2016). The inclusion of stakeholders and cultural ecosystem services in land management trade-off decisions using an ecosystem services approach. *Landscape Ecology*, **31**(3), 533–545.

Davis, K. F., Chhatre, A., Rao, N. D., Singh, D., Ghosh-Jerath, S., Mridul, A., Poblete-Cazenave, M., Pradhan, N., and DeFries, R. (2019). Assessing the sustainability of post-Green Revolution cereals in India. *Proceedings of the National Academy of Sciences of the United States of America*, **116**(50), 25034–25041.

de Araujo Barbosa, C. C., Atkinson, P. M., and Dearing, J. A. (2015). Remote sensing of ecosystem services: A systematic review. *Ecological Indicators*, **52**, 430–443.

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L. C., ten Brink, P., and van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, **1**(1), 50–61.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L., and Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, **7**(3), 260–272.

de Lange, E., Milner-Gulland, E. J., and Keane, A. (2019). Improving Environmental Interventions by Understanding Information Flows. *Trends in Ecology and Evolution*. doi:10.1016/j.tree.2019.06.007

Deguines, N., Jono, C., Baude, M., Henry, M., Julliard, R., and Fontaine, C. (2014). Large-scale trade-off between agricultural intensification and crop pollination services. *Frontiers in Ecology and the Environment*, **12**(4), 212–217.

Dehaan, R. ., and Taylor, G. . (2002). Field-derived spectra of salinized soils and vegetation as indicators of irrigation-induced soil salinization. *Remote Sensing of Environment*, **80**(3), 406–417.

Derissen, S., and Latacz-Lohmann, U. (2013). What are PES? A review of definitions and an extension. *Ecosystem Services*, **6**, 12–15.

Deverka, P. A., Lavallee, D. C., Desai, P. J., Esmail, L. C., Ramsey, S. D., Veenstra, D. L., and Tunis, S. R. (2012). Stakeholder participation in comparative effectiveness research: Defining a framework for effective engagement. *Journal of Comparative Effectiveness Research*, **1**(2). doi:10.2217/cer.12.7

Dick, J., Turkelboom, F., Woods, H., Iniesta-Arandia, I., Primmer, E., Saarela, S. R., Bezák, P., Mederly, P., Leone, M., Verheyden, W., Kelemen, E., Hauck, J., Andrews, C., Antunes, P., Aszalós, R., Baró, F., Barton, D. N., … Zulian, G. (2018). Stakeholders' perspectives on the operationalisation of the ecosystem service concept: Results from 27 case studies. *Ecosystem Services*, **29**. doi:10.1016/j.ecoser.2017.09.015

Dong, C., Schoups, G., and Giesen, N. van de. (2012). Scenario development for decision-making in water resources planning and management: A review. *Technological Forecasting and Social Change*, **2**(September), 928–931.

Donner, S. D., and Kucharik, C. J. (2008). Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River. *Proceedings of the National Academy of*

Sciences of the United States of America, **105**(11), 4513–4518.

Durance, I., Bruford, M. W., Chalmers, R., Chappell, N. A., Christie, M., Cosby, B. J., Noble, D., Ormerod, S. J., Prosser, H., Weightman, A., and Woodward, G. (2016). The Challenges of Linking Ecosystem Services to Biodiversity: Lessons from a Large-Scale Freshwater Study. In *Advances in Ecological Research*, Vol. 54, pp. 87–134.

Duvernoy, I., Zambon, I., Sateriano, A., and Salvati, L. (2018). Pictures from the other side of the fringe: Urban growth and peri-urban agriculture in a post-industrial city (Toulouse, France). *Journal of Rural Studies*, **57**, 25–35.

Egoh, B. N., Reyers, B., Rouget, M., Bode, M., and Richardson, D. M. (2009). Spatial congruence between biodiversity and ecosystem services in South Africa. *Biological Conservation*, **142**(3), 553–562.

Egoh, B., Rouget, M., Reyers, B., Knight, A. T., Cowling, R. M., van Jaarsveld, A. S., and Welz, A. (2007). Integrating ecosystem services into conservation assessments: A review. *Ecological Economics*, **63**(4), 714–721.

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D., and Gaston, K. J. (2010). The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology*, **47**(2), 377–385.

Eliasson, J. (2015). The rising pressure of global water shortages. *Nature*, p. 6.

Elliott, M., Burdon, D., Hemingway, K. L., and Apitz, S. E. (2007). Estuarine, coastal and marine ecosystem restoration: Confusing management and science - A revision of concepts. *Estuarine, Coastal and Shelf Science*. doi:10.1016/j.ecss.2007.05.034

Elosegi, A., and Sabater, S. (2013). Effects of hydromorphological impacts on river ecosystem functioning: A review and suggestions for assessing ecological impacts. *Hydrobiologia*, pp. 129–143.

Endo, A., Tsurita, I., Burnett, K., and Orencio, P. M. (2017). A review of the current state of research on the water, energy, and food nexus. *Journal of Hydrology: Regional Studies*, **11**. doi:10.1016/j.ejrh.2015.11.010

Engel, S., Pagiola, S., and Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, **65**(4), 663–674.

Engel, V., Jobbagy, E. G., Stieglitz, M., Williams, M., and Jackson, R. B. (2005). Hydrological consequences of Eucalyptus afforestation in the Argentine Pampas. *Water Resources Research*, **41**(10). doi:10.1029/2004WR003761

Ens, B., Goodwin, S., Prouzeau, A., Anderson, F., Wang, F. Y., Gratzl, S., Lucarelli, Z., Moyle, B., Smiley, J., and Dwyer, T. (2021). Uplift: A Tangible and Immersive Tabletop System for Casual Collaborative Visual Analytics. *IEEE Transactions on Visualization and Computer Graphics*, **27**(2). doi:10.1109/TVCG.2020.3030334

Erdem, S., Rigby, D., and Wossink, A. (2012). Using best-worst scaling to explore perceptions of relative responsibility for ensuring food safety. *Food Policy*, **37**(6). doi:10.1016/j.foodpol.2012.07.010

Ervin, B. D., Brown, D., Chang, H., Dujon, V., Granek, E., Shandas, V., and Yeakley, A. (2012). Growing Cities Depend on Ecosystem Services. *Solutions*, **2**(6), 1–11.

Esri. (2016). ArcMap 10.4.1. *ESRI*.

Etienne, M., du Toit, D. R., and Pollard, S. (2011). ARDI: A co-construction method for participatory modeling in natural resources management. *Ecology and Society*, **16**(1).

European Environmental Agency. (2018). *European waters Assessment of status and pressures 2018*. *Parents and Children Communicating with Society: Managing Relationships Outside of Home*.

Evans, A. E., Mateo-Sagasta, J., Qadir, M., Boelee, E., and Ippolito, A. (2019). Agricultural water pollution: key knowledge gaps and research needs. *Current Opinion in Environmental Sustainability*, pp. 20–27.

Evans, A., Strezov, V., and Evans, T. (2010). Comparing the sustainability parameters of renewable, nuclear and fossil fuel electricity generation technologies. *World Energy Conference*.

Ewing, P. M., and Runck, B. C. (2015). Optimizing nitrogen rates in the midwestern United States for maximum ecosystem value. *Ecology and Society*, **20**(1). doi:10.5751/ES-06767-200118

Farkas, K., Green, E., Rigby, D., Cross, P., Tyrrel, S., Malham, S. K., and Jones, D. L. (2021). Investigating awareness, fear and control associated with norovirus and other pathogens and pollutants using best–worst scaling. *Scientific Reports*, **11**(1). doi:10.1038/s41598-021-90704-7

Fears, R., Canales, C., ter Meulen, V., and von Braun, J. (2019). Transforming food systems to deliver healthy, sustainable diets—the view from the world's science academies. *The Lancet Planetary Health*, pp. e163–e165.

Fedreheim, G. E., and Blanco, E. (2017). Co-management of protected areas to alleviate conservation conflicts: Experiences in Norway. *International Journal of the Commons*, **11**(2), 754–773.

Finn, J. A., Bartolini, F., Bourke, D., Kurz, I., and Viaggi, D. (2009). Ex post environmental evaluation of agri-environment schemes using experts' judgements and multicriteria analysis. *Journal of Environmental Planning and Management*, **52**(5). doi:10.1080/09640560902958438

Fish, R. D., Winter, M., Oliver, D. M., Chadwick, D. R., Hodgson, C. J., and Heathwaite, A. L. (2013). Employing the citizens' jury technique to elicit reasoned public judgments about environmental risk: insights from an inquiry into the governance of microbial water pollution. *Journal of Environmental Planning and Management*, **57**(2), 233–253.

Fish, R., Winter, M., Oliver, D. M., Chadwick, D., Selfa, T., Heathwaite, A. L., and Hodgson, C. (2009). Unruly pathogens: eliciting values for environmental risk in the context of heterogeneous expert knowledge. *Environmental Science and Policy*, **12**(3), 281–296.

Fleming, G., and MacDougall, K. (2008). *EnviroCentre Report No: 3329 "River Spey Abstractions" to the Spey Fishery Board*. Retrieved from https://www.speyfisheryboard.com/docs/SFB_Final_Envirocentre_Report.pdf

Foley, J. (2011). Can We Feed the World & Sustain the Planet? *Scientific American*, **305**(November), 60–65.

Foley, J., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E., Kucharik, C. J., Monfreda, C., Patz, J., Prentice, I. C., … Snyder, P. K. (2005). Global consequences of land use. *Science*, **309**(5734), 570–4.

Francesconi, W., Srinivasan, R., Perez-Miñana, E., Willcock, S. P., and Quintero, M. (2016). Using the Soil and Water Assessment Tool (SWAT) to model Ecosystem Services: A Systematic Review. *Journal of Hydrology*, **535**, 625–636.

Freeman, L., Everett, M., and Borgatti, S. (2015). UCINET 6.

Freeman, R. E. (2015). *Strategic management: A stakeholder approach*. *Strategic Management: A Stakeholder Approach*. doi:10.1017/CBO9781139192675

Fu, B., Zhang, L., Xu, Z., Zhao, Y., Wei, Y., and Skinner, D. (2015). Ecosystem services in changing land use. *Journal of Soils and Sediments*, **15**(4), 833–843.

Gain, A. K., Hossain, S., Benson, D., Di Baldassarre, G., Giupponi, C., and Huq, N. (2021). Social-ecological system approaches for water resources management. *International Journal of Sustainable Development and World Ecology*, **28**(2). doi:10.1080/13504509.2020.1780647

Galafassi, D., Daw, T. M., Munyi, L., Brown, K., Barnaud, C., and Fazey, I. (2017). Learning about social-ecological trade-offs. *Ecology and Society*, **22**(1). doi:10.5751/ES-08920-220102

Galler, C., Albert, C., and von Haaren, C. (2016). From regional environmental planning to implementation: Paths and challenges of integrating ecosystem services. *Ecosystem Services*, **18**. doi:10.1016/j.ecoser.2016.02.031

Garcia-Llorente, M., Iniesta-arandia, I., Willaarts, B. A., Harrison, P. A., Berry, P., Bayo, M. del M., Castro, A. J., Montes, C., Martín-López, B., and Castro, A. J. (2015). Biophysical and sociocultural factors underlying spatial trade-offs of ecosystem services in semiarid watersheds Biophysical and sociocultural factors underlying spatial trade-offs of ecosystem services in semiarid watersheds. *Ecology and Society*, **20**(3), 39.

García-Nieto, A. P., García-Llorente, M., Iniesta-Arandia, I., and Martín-López, B. (2013). Mapping forest ecosystem services: From providing units to beneficiaries. *Ecosystem Services*, **4**, 126–138.

García-Nieto, A. P., Quintas-Soriano, C., García-Llorente, M., Palomo, I., Montes, C., and Martín-López, B. (2015). Collaborative mapping of ecosystem services: The role of stakeholders' profiles. *Ecosystem Services*, **13**, 141–152.

Gerner, N. V., Nafo, I., Winking, C., Wencki, K., Strehl, C., Wortberg, T., Niemann, A., Anzaldua, G., Lago, M., and Birk, S. (2018). Large-scale river restoration pays off: A case study of ecosystem service valuation for the Emscher restoration generation project. *Ecosystem Services*, **30**. doi:10.1016/j.ecoser.2018.03.020

Gilvear, D. J., Spray, C. J., and Casas-Mulet, R. (2013). River rehabilitation for the delivery of multiple ecosystem services at the river network scale. *Journal of Environmental Management*, **126**, 30–43.

Giuffrida, N., Le Pira, M., Inturri, G., and Ignaccolo, M. (2019). Mapping with stakeholders: An overview of public participatory GIS and VGI in transport decision-making. *ISPRS International Journal of Geo-Information*, **8**(4). doi:10.3390/ijgi8040198

Gleick, P. H. (2003). Global Freshwater Resources : Soft-Path Solutions for the 21st Century. *Science*, **302**, 1524–1528.

Gober, P. (2018). *Building resilience for uncertain water futures*. *Building Resilience for Uncertain Water Futures*. doi:10.1007/978-3-319-71234-5

Goltz, J. von der, Dar, A., Fishman, R., Mueller, N. D., Barnwal, P., and McCord, G. C. (2020). Health Impacts of the Green Revolution: Evidence from 600,000 births across the Developing World. *Journal of Health Economics*, **74**. doi:10.1016/J.JHEALECO.2020.102373

Gomez-Baggethun, E., and Ruiz-Perez, M. (2011). Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography*, 1–16.

Gordon, L. J., Peterson, G. D., and Bennett, E. M. (2008). Agricultural modifications of hydrological flows create ecological surprises. *Trends in Ecology and Evolution*, pp. 211–219.

Graversgaard, M., Jacobsen, B. H., Kjeldsen, C., and Dalgaard, T. (2017). Stakeholder engagement and knowledge co-creation in water planning: Can public participation increase cost-effectiveness? *Water (Switzerland)*, **9**(3). doi:10.3390/w9030191

Green, P. A., Vörösmarty, C. J., Harrison, I., Farrell, T., Sáenz, L., and Fekete, B. M. (2015). Freshwater ecosystem services supporting humans: Pivoting from water crisis to water solutions. *Global Environmental Change*, **34**(SEPTEMBER 2015), 108–118.

Greig, B., and Rathjen, J. (2021). Promoting sustainable industrial water use: Scotland's "Hydro Nation" at home and abroad. In *Sustainable Industrial Water Use: Perspectives, Incentives, and Tools*. doi:10.2166/9781789060676_0395

Groot, J. C. J., Yalew, S. G., and Rossing, W. A. H. (2018). Exploring ecosystem services trade-offs in agricultural landscapes with a multi-objective programming approach. *Landscape and Urban Planning*, **172**, 29–36.

Hack, J. (2015). Application of payments for hydrological ecosystem services to solve problems of fit and interplay in integrated water resources management. *Water International*, **40**(5–6), 929–948.

Hamilton, S. K. (2012). Biogeochemical time lags may delay responses of streams to ecological restoration. *Freshwater Biology*, **57**(SUPPL. 1), 43–57.

Han, Z., Song, W., Deng, X., and Xu, X. (2017). Trade-offs and synergies in ecosystem service within the three-rivers Headwater region, China. *Water (Switzerland)*, **9**(8). doi:10.3390/w9080588

Harris, G. P., and Heathwaite, A. L. (2012). Why is achieving good ecological outcomes in rivers so difficult? *Freshwater Biology*, **57**(SUPPL. 1). doi:10.1111/j.1365-2427.2011.02640.x

Harrison, P. A., Vandewalle, M., Sykes, M. T., Berry, P. M., Bugter, R., de Bello, F., Feld, C. K., Grandin, U., Harrington, R., Haslett, J. R., Jongman, R. H. G., Luck, G. W., da Silva, P. M., Moora, M., Settele, J., Sousa, J. P., and Zobel, M. (2010). Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, **19**(10), 2791–2821.

Hart, S. L., and Sharma, S. (2004). Engaging fringe stakeholders for competitive imagination. *IEEE Engineering Management Review*. doi:10.1109/EMR.2004.25105

Haworth, B., Whittaker, J., and Bruce, E. (2016). Assessing the application and value of participatory mapping for community bushfire preparation. *Applied Geography*, **76**, 115–127.

He, Z., Huang, D., Zhang, C., and Fang, J. (2018). Toward a stakeholder perspective on social stability risk of large hydraulic engineering projects in China: A social network analysis. *Sustainability (Switzerland)*, **10**(4). doi:10.3390/su10041223

Heathwaite, A. L. (2010). Multiple stressors on water availability at global to catchment scales: Understanding human impact on nutrient cycles to protect water quality and water availability in the long term. *Freshwater Biology*, **55**(SUPPL. 1), 241–257.

Heidrich, O., Harvey, J., and Tollin, N. (2009). Stakeholder analysis for industrial waste management systems. *Waste Management*. doi:10.1016/j.wasman.2008.04.013

Hein, L., van Koppen, K., de Groot, R. S., and van Ierland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics*, **57**(2), 209–228.

Hewett, C. J. M., Quinn, P. F., Heathwaite, A. L., Doyle, A., Burke, S., Whitehead, P. G., and Lerner, D. N. (2009). A multi-scale framework for strategic management of diffuse pollution.

Environmental Modelling & Software, **24**(1), 74–85.

Hijmans, R. J. (2015). raster: Geographic Data Analysis and Modeling. R package version 2.5-2.

Hogeboom, R. J., Borsje, B. W., Deribe, M. M., van der Meer, F. D., Mehvar, S., Meyer, M. A., Özerol, G., Hoekstra, A. Y., and Nelson, A. D. (2021). Resilience Meets the Water–Energy–Food Nexus: Mapping the Research Landscape. *Frontiers in Environmental Science*. doi:10.3389/fenvs.2021.630395

Holifield, R., and Williams, K. C. (2019). Recruiting, integrating, and sustaining stakeholder participation in environmental management: A case study from the Great Lakes Areas of Concern. *Journal of Environmental Management*, **230**, 422–433.

Holzkämper, A., Kumar, V., Surridge, B. W. J., Paetzold, A., and Lerner, D. N. (2012). Bringing diverse knowledge sources together e A meta-model for supporting integrated catchment management. *Journal of Environmental Management*, **96**(1), 116–127.

Honey-Rosés, J., Acuña, V., Bardina, M., Brozović, N., Marcé, R., Munné, A., Sabater, S., Termes, M., Valero, F., Vega, À., and Schneider, D. W. (2013). Examining the Demand for Ecosystem Services: The Value of Stream Restoration for Drinking Water Treatment Managers in the Llobregat River, Spain. *Ecological Economics*, **90**, 196–205.

Horning, D., Bauer, B. O., and Cohen, S. J. (2016). Missing bridges: Social network (dis)connectivity in water governance. *Utilities Policy*, **43**, 59–70.

Hou, Y., Burkhard, B., and Müller, F. (2013). Uncertainties in landscape analysis and ecosystem service assessment. *Journal of Environmental Management*, **127**, S117–S131.

Howe, C., Suich, H., Vira, B., and Mace, G. M. (2014). Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change*, **28**, 263–275.

Hughes, C. J., Sinclair, F. L., Pagella, T., and Roberts, J. C. (2011). Calibrating the kinect with a 3D projector to create a tangible tabletop interface. In *VTT Symposium (Valtion Teknillinen Tutkimuskeskus)*.

Huitema, D., Cornelisse, C., and Ottow, B. (2010). Is the jury still out? toward greater insight in policy learning in participatory decision processes-the case of dutch citizens' juries on water management in the rhine basin. *Ecology and Society*, **15**(1). doi:16

Hummel, S. (2017). Relative water scarcity and country relations along cross-boundary rivers: Evidence from the Aral Sea basin. *International Studies Quarterly*, **61**(4), 795–808.

IBM. (2012). IBM SPSS Advanced Statistics 23. *Ibm*, 184.

IBM. (2021). IBM SPSS Advanced Statistics 28.

IEA (International Energy Agency). (2018). World Energy Outlook 2018: Highlights. *International Energy Agency*, **1**, 1–661.

Jacobs, S., Burkhard, B., Van Daele, T., Staes, J., and Schneiders, A. (2015). 'The Matrix Reloaded': A review of expert knowledge use for mapping ecosystem services. *Use of Ecological Indicators in Models*, **295**, 21–30.

Jarvie, H. P., Sharpley, A. N., Spears, B., Buda, A. R., May, L., and Kleinman, P. J. A. (2013). Water quality remediation faces unprecedented challenges from "legacy Phosphorus." *Environmental Science and Technology*, pp. 8997–8998.

Jensen, D., Baird, T., and Blank, G. (2019). New landscapes of conflict: land-use competition at the urban–rural fringe. *Landscape Research*, **44**(4), 418–429.

John, D. A., and Babu, G. R. (2021, February 22). Lessons From the Aftermaths of Green Revolution on Food System and Health. *Frontiers in Sustainable Food Systems*, Europe PMC Funders, p. 644559.

Johnson, R. K., Angeler, D. G., Hallstan, S., Sandin, L., and McKie, B. G. (2017). Decomposing multiple pressure effects on invertebrate assemblages of boreal streams. *Ecological Indicators*, **77**, 293–303.

Joint Nature Conservation Committee. (2013). Natura 2000 Standard Data Form. *Quality*, 1–3.

Jones, A. K., Jones, D. L., Edwards-Jones, G., and Cross, P. (2013). Informing decision making in agricultural greenhouse gas mitigation policy: A Best-Worst Scaling survey of expert and farmer opinion in the sheep industry. *Environmental Science and Policy*, **29**. doi:10.1016/j.envsci.2013.02.003

Jones, A. R., Doubleday, Z. A., Prowse, T. A. A., Wiltshire, K. H., Deveney, M. R., Ward, T., Scrivens, S. L., Cassey, P., O'Connell, L. G., and Gillanders, B. M. (2018). Capturing expert uncertainty in spatial cumulative impact assessments /631/158 /631/158/2445 article. *Scientific Reports*, **8**(1). doi:10.1038/s41598-018-19354-6

Jones, P. (2018). Contexts of Co-creation: Designing with System Stakeholders. In J. P. & K. K., eds., *Systemic Design. Translational Systems Sciences*, 8th edn, Tokyo: Springer, pp. 3–52.

Joseph, L. N., Maloney, R. F., and Possingham, H. P. (2009). Optimal allocation of resources among threatened species: a project prioritization protocol. *Conservation Biology*, **23**(2), 328–338.

Kahan, D. M., Wittlin, M., and Peters, E. (2012). The polarizing impact of science literacy and numeracy on perceived climate change risks. *Nature Climate Change*, 732–735.

Karabulut, A., Egoh, B. N., Lanzanova, D., Grizzetti, B., Bidoglio, G., Pagliero, L., Bouraoui, F., Aloe, A., Reynaud, A., Maes, J., Vandecasteele, I., and Mubareka, S. (2016). Mapping water provisioning services to support the ecosystem-water-food-energy nexus in the Danube river basin. *Ecosystem Services*, **17**, 278–292.

Karimi, A., and Brown, G. (2017). Assessing multiple approaches for modelling land-use conflict potential from participatory mapping data. *Land Use Policy*, **67**, 253–267.

Karp, D. S., Mendenhall, C. D., Sandí, R. F., Chaumont, N., Ehrlich, P. R., Hadly, E. A., and Daily, G. C. (2013). Forest bolsters bird abundance, pest control and coffee yield. *Ecology Letters*, pp. 1339–1347.

Karpouzoglou, T., Dewulf, A., and Clark, J. (2016). Advancing adaptive governance of social-ecological systems through theoretical multiplicity. *Environmental Science and Policy*, pp. 1–9.

Kasina, J. M., Mburu, J., Kraemer, M., and Holm-Mueller, K. (2009). Economic benefit of crop pollination by bees: a case of Kakamega small-holder farming in western Kenya. *Journal of Economic Entomology*, **102**(2), 467–473.

Kaye-Zwiebel, E., and King, E. (2014). Kenyan pastoralist societies in transition: varying perceptions of the value of ecosystem services. *Ecology and Society*, **19**(3), 17.

Keeler, B. L., Polasky, S., Brauman, K. A., Johnson, K. A., Finlay, J. C., O'Neill, A., Kovacs, K., and Dalzell, B. (2012). Linking water quality and well-being for improved assessment and valuation of ecosystem services. *Proceedings of the National Academy of Sciences*, **109**(45), 18619–18624.

Keeler, B. L., Wood, S. A., Polasky, S., Kling, C., Filstrup, C. T., and Downing, J. A. (2015). Recreational demand for clean water: Evidence from geotagged photographs by visitors to lakes. *Frontiers in Ecology and the Environment*, **13**(2), 76–81.

Kenyon, W., Nevin, C., and Hanley, N. (2003). Enhancing Environmental Decision-making Using Citizens' Juries. *Local Environment*, **8**(2), 221–232.

Khan, I., Hou, F., and Le, H. P. (2021). The impact of natural resources, energy consumption, and population growth on environmental quality: Fresh evidence from the United States of America. *Science of the Total Environment*, **754**. doi:10.1016/j.scitotenv.2020.142222

King, E., Cavender-Bares, J., Balvanera, P., Mwampamba, T. H., and Polasky, S. (2015). Trade-offs in ecosystem services and varying stakeholder preferences: Evaluating conflicts, obstacles, and opportunities. *Ecology and Society*, **20**(3). doi:10.5751/ES-07822-200325

Kirchner, M., Schmidt, J., Kindermann, G., Kulmer, V., Mitter, H., Prettenthaler, F., Rüdisser, J., Schauppenlehner, T., Schönhart, M., Strauss, F., Tappeiner, U., Tasser, E., and Schmid, E. (2015). Ecosystem services and economic development in Austrian agricultural landscapes — The impact of policy and climate change scenarios on trade-offs and synergies. *Ecological Economics*, **109**, 161–174.

Kirschke, S., Zhang, L., and Meyer, K. (2018). Decoding the Wickedness of Resource Nexus Problems—Examples from Water-Soil Nexus Problems in China. *Resources*, **7**(4), 67.

Klaar, M. J., Carver, S., and Kay, P. (2020). Land management in a post-Brexit UK: An opportunity for integrated catchment management to deliver multiple benefits? *Wiley Interdisciplinary Reviews: Water*, **7**(5). doi:10.1002/wat2.1479

Klæboe, R., and Sundfør, H. B. (2016). Windmill noise annoyance, visual aesthetics, and attitudes towards renewable energy sources. *International Journal of Environmental Research and Public Health*, **13**(8). doi:10.3390/ijerph13080746

Klain, S. C., and Chan, K. M. A. (2012). Navigating coastal values: Participatory mapping of ecosystem services for spatial planning. *Ecological Economics*, **82**, 104–113.

Klain, S. C., Satterfield, T. a., and Chan, K. M. a. (2014). What matters and why? Ecosystem services and their bundled qualities. *Ecological Economics*, **107**, 310–320.

Knight, A. T., Cowling, R. M., and Campbell, B. M. (2006). An operational model for implementing conservation action. *Conservation Biology*, pp. 408–419.

Knol, A. B., Slottje, P., Van Der Sluijs, J. P., and Lebret, E. (2010). The use of expert elicitation in environmental health impact assessment: A seven step procedure. *Environmental Health: A Global Access Science Source*, **9**(1). doi:10.1186/1476-069X-9-19

Knook, J., Dynes, R., Pinxterhuis, I., de Klein, C. A. M., Eory, V., Brander, M., and Moran, D. (2020). Policy and Practice Certainty for Effective Uptake of Diffuse Pollution Practices in A Light-Touch Regulated Country. *Environmental Management*, **65**(2). doi:10.1007/s00267-019-01242-y

Koch, E. W., Barbier, E. B., Silliman, B. R., Reed, D. J., Perillo, G. M. E., Hacker, S. D., Granek, E. F., Primavera, J. H., Muthiga, N., Polasky, S., Halpern, B. S., Kennedy, C. J., Kappel, C. V., and Wolanski, E. (2009). Non-linearity in ecosystem services: Temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment*, pp. 29–37.

Kochskämper, E., Challies, E., Newig, J., and Jager, N. W. (2016). Participation for effective environmental governance? Evidence from Water Framework Directive implementation in Germany, Spain and the United Kingdom. *Journal of Environmental Management*, **181**, 737–

748.

Köpsel, V., de Moura Kiipper, G., and Peck, M. A. (2021). Stakeholder engagement vs. social distancing—how does the Covid-19 pandemic affect participatory research in EU marine science projects? *Maritime Studies*, **20**(2). doi:10.1007/s40152-021-00223-4

Kosoy, N., and Corbera, E. (2010). Payments for ecosystem services as commodity fetishism. *Ecological Economics*, **69**(6), 1228–1236.

Kovács, E., Kelemen, E., Kalóczkai, Á., Margóczi, K., Pataki, G., Gébert, J., Málovics, G., Balázs, B., Roboz, Á., Krasznai Kovács, E., and Mihók, B. (2015). Understanding the links between ecosystem service trade-offs and conflicts in protected areas. *Ecosystem Services*, **12**, 117–127.

Krackhardt, D. (1987). Cognitive social structures. *Social Networks*, **9**(2). doi:10.1016/0378-8733(87)90009-8

Krueger, T., Page, T., Hubacek, K., Smith, L., and Hiscock, K. (2012). The role of expert opinion in environmental modelling. *Environmental Modelling and Software*, **36**. doi:10.1016/j.envsoft.2012.01.011

Kuhlicke, C., Steinführer, A., Begg, C., Bianchizza, C., Bründl, M., Buchecker, M., De Marchi, B., Di Masso Tarditti, M., Höppner, C., Komac, B., Lemkow, L., Luther, J., McCarthy, S., Pellizzoni, L., Renn, O., Scolobig, A., Supramaniam, M., … Faulkner, H. (2011). Perspectives on social capacity building for natural hazards: outlining an emerging field of research and practice in Europe. *Environmental Science & Policy*, **14**(7), 804–814.

Kujala, J., Sachs, S., Leinonen, H., Heikkinen, A., and Daniel, L. (2022). Stakeholder Engagement: Past, Present, and Future. *Business & Society*, 1–61.

Lagerkvist, C. J., Okello, J., and Karanja, N. (2012). Anchored vs. relative best-worst scaling and latent class vs. hierarchical Bayesian analysis of best-worst choice data: Investigating the importance of food quality attributes in a developing country. *Food Quality and Preference*, **25**(1). doi:10.1016/j.foodqual.2012.01.002

Lamarque, P., Quétier, F., and Lavorel, S. (2011). The diversity of the ecosystem services concept and its implications for their assessment and management. *Comptes Rendus Biologies*, **334**(5–6), 441–449.

Landuyt, D., Lemmens, P., D'hondt, R., Broekx, S., Liekens, I., De Bie, T., Declerck, S. a J., De Meester, L., and Goethals, P. L. M. (2014). An ecosystem service approach to support integrated pond management: a case study using Bayesian belief networks--highlighting opportunities and risks. *Journal of Environmental Management*, **145**, 79–87.

Lane, S. N., Brookes, C. J., Heathwaite, A. L., and Reaney, S. (2006). Surveillant science: Challenges for the management of rural environments emerging from the new generation diffuse pollution models. *Journal of Agricultural Economics*, **57**(2). doi:10.1111/j.1477-9552.2006.00050.x

Lang, Y., and Song, W. (2018). Trade-off analysis of ecosystem services in a mountainous karst area, China. *Water (Switzerland)*, **10**(3), 1–21.

Lautenbach, S., Volk, M., Strauch, M., Whittaker, G., and Seppelt, R. (2013). Optimization-based trade-off analysis of biodiesel crop production for managing an agricultural catchment. *Environmental Modelling and Software*, **48**, 98–112.

Lawler, J. J., Lewis, D. J., Nelson, E., Plantinga, A. J., Polasky, S., Withey, J. C., Helmers, D. P., Martinuzzi, S., Pennington, D., and Radeloff, V. C. (2014). Projected land-use change impacts on ecosystem services in the United States. *Proceedings of the National Academy of Sciences of* *the United States of America*, **111**(20), 7492–7.

Lechner, A. M., Verbrugge, L. N. H., Chelliah, A., Ang, M. L. E., and Raymond, C. M. (2020). Rethinking tourism conflict potential within and between groups using participatory mapping. *Landscape and Urban Planning*, **203**, 103902.

Lee, H., and Lautenbach, S. (2016). A quantitative review of relationships between ecosystem services. *Ecological Indicators*, pp. 340–351.

Lee, J. A., Soutar, G., and Louviere, J. (2008). The best-worst scaling approach: An alternative to Schwartz's values survey. *Journal of Personality Assessment*, **90**(4). doi:10.1080/00223890802107925

Lees, A. C., Peres, C. A., Fearnside, P. M., Schneider, M., and Zuanon, J. A. S. (2016). Hydropower and the future of Amazonian biodiversity. *Biodiversity and Conservation*, pp. 451–466.

Lemos, M. C., Arnott, J. C., Ardoin, N. M., Baja, K., Bednarek, A. T., Dewulf, A., Fieseler, C., Goodrich, K. A., Jagannathan, K., Klenk, N., Mach, K. J., Meadow, A. M., Meyer, R., Moss, R., Nichols, L., Sjostrom, K. D., Stults, M., … Wyborn, C. (2018). To co-produce or not to co-produce. *Nature Sustainability*. doi:10.1038/s41893-018-0191-0

Lerner, D. N., Kumar, V., Holzk, A., Surridge, B. W. J., and Harris, B. (2011). Challenges in developing an integrated catchment management model, **25**, 345–354.

Lerner, D. N., Kumar, V., Holzkämper, A., Surridge, B. W. J., and Harris, B. (2010). Challenges in developing an integrated catchment management model. *Water and Environment Journal*, Online Early View.

Lerner, D. N., and Zheng, C. (2011). Integrated catchment management: path to enlightenment. *Hydrological Processes*, **25**(16), 2635–2640.

Lester, S. E., Costello, C., Halpern, B. S., Gaines, S. D., White, C., and Barth, J. A. (2013). Evaluating tradeoffs among ecosystem services to inform marine spatial planning. *Marine Policy*, **38**, 80–89.

Levin, N., Tulloch, A. I. T., Gordon, A., Mazor, T., Bunnefeld, N., and Kark, S. (2013). Incorporating Socioeconomic and Political Drivers of International Collaboration into Marine Conservation Planning. *BioScience*, **63**(7), 547–563.

Lienert, J., Schnetzer, F., and Ingold, K. (2013). Stakeholder analysis combined with social network analysis provides fine-grained insights into water infrastructure planning processes. *Journal of Environmental Management*. doi:10.1016/j.jenvman.2013.03.052

Likens, G. E., Walker, K. F., Davies, P. E., Brookes, J., Olley, J., Young, W. J., Thoms, M. C., Lake, P. S., Gawne, B., Davis, J., Arthington, A. H., Thompson, R., and Oliver, R. L. (2009). Ecosystem science: Toward a new paradigm for managing Australia's inland aquatic ecosystems. *Marine and Freshwater Research*, pp. 271–279.

Lin, S., Wu, R., Yang, F., Wang, J., and Wu, W. (2018). Spatial trade-offs and synergies among ecosystem services within a global biodiversity hotspot. *Ecological Indicators*, **84**, 371–381.

Lintern, A., McPhillips, L., Winfrey, B., Duncan, J., and Grady, C. (2020). Best Management Practices for Diffuse Nutrient Pollution: Wicked Problems across Urban and Agricultural Watersheds. *Environmental Science and Technology*. doi:10.1021/acs.est.9b07511

Liquete, C., Maes, J., Notte, A. La, and Bidoglio, G. (2012). Securing water as a resource for society: an ecosystem services perspective. *Ecohydrology & Hydrobiology*, **11**(3–4), 247–259.

Liu, S., Costanza, R., Farber, S., and Troy, A. (2010). Valuing ecosystem services: Theory, practice, and the need for a transdisciplinary synthesis, **1185**, 54–78.

Llopis-Albert, C., Merigó, J. M., Xu, Y., and Liao, H. (2017). Application of Fuzzy Set/Qualitative Comparative Analysis to Public Participation Projects in Support of the EU Water Framework Directive. *Water Environment Research*, **90**(1). doi:10.2175/106143017x15054988926550

Loureiro, M. L., and Dominguez Arcos, F. (2012). Applying Best-Worst Scaling in a stated preference analysis of forest management programs. *Journal of Forest Economics*, **18**(4), 381–394.

Louviere, J., Lings, I., Islam, T., Gudergan, S., and Flynn, T. (2013). An introduction to the application of (case 1) best-worst scaling in marketing research. *International Journal of Research in Marketing*, **30**(3). doi:10.1016/j.ijresmar.2012.10.002

Love, S., Paine, M., Melland, A., and Gourley, C. (2006). *Research or extension? Scientists participating in collaborative catchment management*. Retrieved from http://www.regional.org.au/au/apen/2006/refereed/3/3096_loves.htm

Lu, N., Fu, B., Jin, T., and Chang, R. (2014). Trade-off analyses of multiple ecosystem services by plantations along a precipitation gradient across Loess Plateau landscapes. *Landscape Ecology*, **29**(10), 1697–1708.

Luzi, S., Abdelmoghny Hamouda, M., Sigrist, F., and Tauchnitz, E. (2008). Water policy networks in Egypt and Ethiopia. *Journal of Environment and Development*, **17**(3), 238–268.

Ma, S., Duggan, J. M., Eichelberger, B. A., McNally, B. W., Foster, J. R., Pepi, E., Conte, M. N., Daily, G. C., and Ziv, G. (2016). Valuation of ecosystem services to inform management of multiple-use landscapes. *Ecosystem Services*, **19**, 6–18.

Ma, W., Li, X., and Wang, X. (2021). Water Saving Management Contract, identification and ranking of risks based on life cycle and best-worst method. *Journal of Cleaner Production*, **306**. doi:10.1016/j.jclepro.2021.127153

Mach, K. J., Mastrandrea, M. D., Freeman, P. T., and Field, C. B. (2017). Unleashing expert judgment in assessment. *Global Environmental Change*, **44**, 1–14.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P., Grizzetti, B., Drakou, E. G., Notte, A. La, Zulian, G., Bouraoui, F., Luisa Paracchini, M., Braat, L., and Bidoglio, G. (2012). Mapping ecosystem services for policy support and decision making in the European Union. *Ecosystem Services*, **1**(1), 31–39.

Mahboubi, P., Parkes, M., Stephen, C., and Chan, H. M. (2015). Using expert informed GIS to locate important marine social-ecological hotspots. *Journal of Environmental Management*, **160**, 342–352.

Malcolm, I. A., Soulsby, C., Hannah, D. M., Bacon, P. J., Youngson, A. F., and Tetzlaff, D. (2008). The influence of riparian woodland on stream temperatures: Implications for the performance of juvenile salmonids. *Hydrological Processes*, **22**(7), 968–979.

Malcolm, N. (2011). Rivers in trust: Stakeholders and delivery of the EU water framework directive. *Proceedings of the Institution of Civil Engineers: Water Management*. doi:10.1680/wama.2011.164.8.433

Manning, P., Van Der Plas, F., Soliveres, S., Allan, E., Maestre, F. T., Mace, G., Whittingham, M. J., and Fischer, M. (2018). Redefining ecosystem multifunctionality. *Nature Ecology and Evolution*. doi:10.1038/s41559-017-0461-7

Martin-Lopez, B., Iniesta-Arandia, I., Garcia-Llorente, M., Palomo, I., Casado-Arzuaga, I., Garcia del

Amo, D., Gomez-Baggethun, E., Oteros-Rozas, E., Palacios-Agendez, I., Willaarts, B., Gonzalez, J. A., Santos-Martin, F., Onaindia, M., MLopez-Santiago, C., and Montes, C. (2012). Uncovering ecosystem services bundles through social preferences. *PLoS One*, **7**(6), e38970.

Martin-Ortega, J., Jorda-Capdevila, D., Glenk, K., and Holstead, K. L. (2015). What defines ecosystem services-based approaches? In J. Martin-Ortega, R. C. Ferrier, I. J. Gordon, & S. Khan, eds., *Water Ecosystem Services: A Global Perspective*, Cambridge: Cambridge University Press, pp. 3–13.

Martin, T. G., Burgman, M. A., Fidler, F., Kuhnert, P. M., Low-Choy, S., Mcbride, M., and Mengersen, K. (2012). Eliciting Expert Knowledge in Conservation Science. *Conservation Biology*, pp. 29–38.

Martínez-Dalmau, J., Berbel, J., and Ordóñez-Fernández, R. (2021). Nitrogen fertilization. A review of the risks associated with the inefficiency of its use and policy responses. *Sustainability (Switzerland)*. doi:10.3390/su13105625

Martínez-Harms, M. J., and Balvanera, P. (2012). Methods for mapping ecosystem service supply: a review. *International Journal of Biodiversity Science, Ecosystem Services & Management*, **8**(1–2), 17–25.

Martinez-Haro, M., Beiras, R., Bellas, J., Capela, R., Coelho, J. P., Lopes, I., Moreira-Santos, M., Reis-Henriques, A. M., Ribeiro, R., Santos, M. M., and Marques, J. C. (2015). A review on the ecological quality status assessment in aquatic systems using community based indicators and ecotoxicological tools: What might be the added value of their combination? *Ecological Indicators*, pp. 8–16.

Maskell, L. C., Crowe, A., Dunbar, M. J., Emmett, B., Henrys, P., Keith, A. M., Norton, L. R., Scholefield, P., Clark, D. B., Simpson, I. C., and Smart, S. M. (2013). Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. *Journal of Applied Ecology*, **50**(3), 561–571.

Mason, T. H. E., Pollard, C. R. J., Chimalakonda, D., Guerrero, A. M., Kerr-Smith, C., Milheiras, S. A. G., Roberts, M., R. Ngafack, P., and Bunnefeld, N. (2018). Wicked conflict: Using wicked problem thinking for holistic management of conservation conflict. *Conservation Letters*, **11**(6). doi:10.1111/conl.12460

Mathur, V. N., Price, A. D. F., and Austin, S. (2008). Conceptualizing stakeholder engagement in the context of sustainability and its assessment. *Construction Management and Economics*, **26**(6). doi:10.1080/01446190802061233

Matios, E., and Burney, J. (2017). Ecosystem Services Mapping for Sustainable Agricultural Water Management in California's Central Valley. *Environmental Science & Technology*, **51**(5), 2593–2601.

Matthies, B. D., Kalliokoski, T., Eyvindson, K., Honkela, N., Hukkinen, J. I., Kuusinen, N. J., Räisänen, P., and Valsta, L. T. (2016). Nudging service providers and assessing service trade-offs to reduce the social inefficiencies of payments for ecosystem services schemes. *Environmental Science and Policy*, **55**, 228–237.

Mc Morran, R., and Glass, J. (2013). Buying nature: A review of environmental NGO landownership. In *Lairds, Land and Sustainability: Scottish Perspectives on Upland Management*.

McCall, M. K., and Dunn, C. E. (2012). Geo-information tools for participatory spatial planning: Fulfilling the criteria for "good" governance? *Geoforum*, **43**(1), 81–94.

McCracken, D. I., Cole, L. J., Harrison, W., and Robertson, D. (2012). Improving the Farmland Biodiversity Value of Riparian Buffer Strips: Conflicts and Compromises. *Journal of*

Environmental Quality, **41**(2). doi:10.2134/jeq2010.0532

McEwen, L., Garde-Hansen, J., Holmes, A., Jones, O., and Krause, F. (2017). Sustainable flood memories, lay knowledges and the development of community resilience to future flood risk. *Transactions of the Institute of British Geographers*, **42**(1), 14–28.

McGonigle, D. F., Burke, S. P., Collins, A. L., Gartner, R., Haft, M. R., Harris, R. C., Haygarth, P. M., Hedges, M. C., Hiscock, K. M., and Lovett, A. A. (2014). Developing Demonstration Test Catchments as a platform for transdisciplinary land management research in England and Wales. *Environmental Sciences: Processes and Impacts*, **16**(7). doi:10.1039/c3em00658a

McGonigle, D. F., Harris, R. C., McCamphill, C., Kirk, S., Dils, R., MacDonald, J., and Bailey, S. (2012). Towards a more strategic approach to research to support catchment-based policy approaches to mitigate agricultural water pollution: A UK case-study. *Environmental Science and Policy*, **24**, 4–14.

Mckenzie, A. J., Emery, S. B., Franks, J. R., and Whittingham, M. J. (2013). FORUM: Landscape-scale conservation: Collaborative agri-environment schemes could benefit both biodiversity and ecosystem services, but will farmers be willing to participate? *Journal of Applied Ecology*, **50**(5). doi:10.1111/1365-2664.12122

McVittie, A., Norton, L., Martin-Ortega, J., Siameti, I., Glenk, K., and Aalders, I. (2015). Operationalizing an ecosystem services-based approach using Bayesian Belief Networks: An application to riparian buffer strips. *Ecological Economics*, **110**. doi:10.1016/j.ecolecon.2014.12.004

Melland, A. R., Fenton, O., and Jordan, P. (2018). Effects of agricultural land management changes on surface water quality: A review of meso-scale catchment research. *Environmental Science and Policy*. doi:10.1016/j.envsci.2018.02.011

Mendenhall, E., Hendrix, C., Nyman, E., Roberts, P. M., Hoopes, J. R., Watson, J. R., Lam, V. W. Y., and Sumaila, U. R. (2020). Climate change increases the risk of fisheries conflict. *Marine Policy*, **117**. doi:10.1016/j.marpol.2020.103954

Menzel, S., and Buchecker, M. (2013). Does participatory planning foster the transformation toward more adaptive social-ecological systems? *Ecology and Society*, **18**(1). doi:10.5751/ES-05154-180113

Micha, E., Roberts, W., Ryan, M., O'Donoghue, C., and Daly, K. (2018). A participatory approach for comparing stakeholders' evaluation of P loss mitigation options in a high ecological status river catchment. *Environmental Science and Policy*, **84**, 41–51.

Millennium Ecosystem Assessment (MEA). (2005). Ecosystems and Human Well-being. *Ecosystems*, **5**(281), 1–100.

Mills, J., Chiswell, H., Gaskell, P., Courtney, P., Brockett, B., Cusworth, G., and Lobley, M. (2021). Developing Farm-Level Social Indicators for Agri-Environment Schemes: A Focus on the Agents of Change. *Sustainability*, **13**(14). doi:10.3390/su13147820

Mohamad Ibrahim, I. H., Gilfoyle, L., Reynolds, R., and Voulvoulis, N. (2019). Integrated catchment management for reducing pesticide levels in water: Engaging with stakeholders in East Anglia to tackle metaldehyde. *Science of the Total Environment*, **656**, 1436–1447.

Moore, S. A., Brown, G., Kobryn, H., and Strickland-Munro, J. (2017). Identifying conflict potential in a coastal and marine environment using participatory mapping. *Journal of Environmental Management*, **197**, 706–718.

Morss, R. E., Wilhelmi, O. V, Downton, M. W., and Gruntfest, E. (2005). Flood risk, Uncertainty and Scientific Information for Decision Making. *Bulletin of the American Meterological Society*, 1593–1601.

Mulvaney, K. K., Lee, S., Höök, T. O., and Prokopy, L. S. (2015). Casting a net to better understand fisheries management: An affiliation network analysis of the Great Lakes Fishery Commission. *Marine Policy*, **57**, 120–131.

Munia, H., Guillaume, J. H. A., Mirumachi, N., Porkka, M., Wada, Y., and Kummu, M. (2016). Water stress in global transboundary river basins: significance of upstream water use on downstream stress. *Environmental Research Letters*, **11**(1), 14002.

Muunda, E., Mtimet, N., Schneider, F., Wanyoike, F., Dominguez-Salas, P., and Alonso, S. (2021). Could the new dairy policy affect milk allocation to infants in Kenya? A best-worst scaling approach. *Food Policy*, **101**. doi:10.1016/j.foodpol.2021.102043

Nancarrow, B. E. (2005). When the modeller meets the social scientist or vice-versa. In *MODSIM05 - International Congress on Modelling and Simulation: Advances and Applications for Management and Decision Making, Proceedings*, pp. 38–44.

Navarro-Navarro, L. A., Moreno-Vazquez, J. L., and Scott, C. A. (2017). Social networks for management of water scarcity: Evidence from the San Miguel Watershed, Sonora, Mexico. *Water Alternatives*, **10**(1), 41–64.

Nayak, M. S. D. P., and Narayan, K. A. (2019). Strengths and Weakness of Online Surveys. *IOSR Journal Of Humanities And Social Science*, **24**(5).

Nelson, E. J., and Daily, G. C. (2010). Modelling ecosystem services in terrestrial systems. *F1000 Biology Reports*, **2**, 53.

Nelson, E., Polasky, S., Lewis, D. J., Plantinga, A. J., Lonsdorf, E., White, D., Bael, D., and Lawler, J. J. (2008). Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proceedings of the National Academy of Sciences of the United States of America*, **105**(28), 9471–6.

Nemec, K. T., and Raudsepp-Hearne, C. (2013). The use of geographic information systems to map and assess ecosystem services. *Biodiversity and Conservation*, pp. 1–15.

Noble, M. M., Harasti, D., Pittock, J., and Doran, B. (2019). Understanding the spatial diversity of social uses, dynamics, and conflicts in marine spatial planning. *Journal of Environmental Management*, **246**, 929–940.

Nordén, A., Coria, J., Jénsson, A. M., Lagergren, F., and Lehsten, V. (2017). Divergence in stakeholders' preferences: Evidence from a choice experiment on forest landscapes preferences in Sweden. *Ecological Economics*, **132**, 179–195.

Nuno, A., Bunnefeld, N., and Milner-Gulland, E. J. (2014). Managing social-ecological systems under uncertainty: Implementation in the real world. *Ecology and Society*, **19**(2). doi:10.5751/ES-06490-190252

O'Sullivan, O. S., Holt, A. R., Warren, P. H., and Evans, K. L. (2017). Optimising UK urban road verge contributions to biodiversity and ecosystem services with cost-effective management. *Journal of Environmental Management*, pp. 162–171.

Ockenden, M. C., Deasy, C., Quinton, J. N., Bailey, A. P., Surridge, B., and Stoate, C. (2012). Evaluation of field wetlands for mitigation of diffuse pollution from agriculture: Sediment retention, cost and effectiveness. *Environmental Science and Policy*, **24**. doi:10.1016/j.envsci.2012.06.003

Ogada, J. O., Krhoda, G. O., Van Der Veen, A., Marani, M., and van Oel, P. R. (2017). Managing resources through stakeholder networks: collaborative water governance for Lake Naivasha basin, Kenya. *Water International*, **42**(3), 271–290.

Oliver, D., Bartie, P., Heathwaite, A., Pschetz, L., and Quilliam, R. (2017). Design of a decision support tool for visualising E. coli risks on agricultural land using a stakeholder-driven approach. *Journal of Environmental Management*, **submitted**.

Oliver, D. M., Fish, R. D., Winter, M., Hodgson, C. J., Heathwaite, a. L., and Chadwick, D. R. (2012). Valuing local knowledge as a source of expert data: Farmer engagement and the design of decision support systems. *Environmental Modelling & Software*, **36**, 76–85.

Orme, B. K. (2010). *Getting Started with Conjoint Analysis: Strategies for Product Design and Pricing Research*. *Research Publishers*.

Orme, B. K., and Chrzan, K. (2006). Adaptive Maximum Difference. *Sawtooth Software Research Paper Serires*, **98382**(360).

Ormerod, S. J., Dobson, M., Hildrew, a. G., and Townsend, C. R. (2010). Multiple stressors in freshwater ecosystems. *Freshwater Biology*, **55**, 1–4.

Ortega-Pacheco, D. V., Lupi, F., and Kaplowitz, M. D. (2009). Payment for Environmental Services: Estimating Demand Within a Tropical Watershed. *Journal of Natural Resources Policy Research*, **1**(2), 189–202.

Ostrom, E. (1990). *Governing the Commons: The Evolution of Institutions for Collective Action*. (J. E. Alt & D. C. North, Eds.), Cambridge: Cambridge University Press.

Oteros-Rozas, E., Martín-López, B., Daw, T. M., Bohensky, E. L., Butler, J. R. A., Hill, R., Martin-Ortega, J., Quinlan, A., Ravera, F., Ruiz-Mallén, I., Thyresson, M., Mistry, J., Palomo, I., Peterson, G. D., Plieninger, T., Waylen, K. A., Beach, D. M., … Vilardy, S. P. (2015). Participatory scenario planning in place-based social-ecological research: Insights and experiences from 23 case studies. *Ecology and Society*, **20**(4). doi:10.5751/ES-07985-200432

Page, T., Heathwaite, A. L., Thompson, L. J., Pope, L., and Willows, R. (2012). Environmental Modelling & Software Eliciting fuzzy distributions from experts for ranking conceptual risk model components. *Environmental Modelling and Software*, **36**, 19–34.

Pahl-Wostl, C., Kabat, P., and Möltgen, J. (2008). *Adaptive and Integrated Water Management: Coping With Complexity and Uncertainty*. *Eos, Transactions American Geophysical Union*, Vol. 89. doi:10.1029/2008EO330009

Paveglio, T. B., Carroll, M. S., Hall, T. E., and Brenkert-Smith, H. (2015). "Put the wet stuff on the hot stuff": The legacy and drivers of conflict surrounding wildfire suppression. *Journal of Rural Studies*, **41**, 72–81.

Pe'er, G., Dicks, L. V, Visconti, P., Arlettaz, R., Báldi, A., Benton, T. G., Collins, S., Dieterich, M., Gregory, R. D., Hartig, F., Henle, K., Hobson, P. R., Kleijn, D., Neumann, R. K., Robijns, T., Schmidt, J., Shwartz, A., … Scott, A. V. (2014). EU agricultural reform fails on biodiversity: extra steps by member states are needed to protect farmed and grassland ecosystems. *Science (Washington)*, **344**(6188), 1090–1092.

Pebesma, E. J., and Bivand, R. S. (2005). Classes and methods for spatial data in R. *R News*, **5**(2).

Pellegrini, E., Bortolini, L., and Defrancesco, E. (2019). Coordination and Participation Boards under the European Water Framework Directive: Different approaches used in some EU countries. *Water (Switzerland)*, **11**(4). doi:10.3390/w11040833

Pendleton, L., Mongruel, R., Beaumont, N., Hooper, T., and Charles, M. (2015). A triage approach to improve the relevance of marine ecosystem services assessments. *Marine Ecology Progress Series*, **530**, 183–193.

Petersen-Perlman, J. D., Veilleux, J. C., and Wolf, A. T. (2017). International water conflict and cooperation: challenges and opportunities. *Water International*, pp. 105–120.

Petts, J., Judith Petts, and Petts, J. (2001). Evaluating the Effectiveness of Deliberative Processes: Waste Management Case-studies. *Journal of Environmental Planning and Management*, **44**(2), 207–226.

Philpot, S., Hipel, K., and Johnson, P. (2019). Identifying Potential Conflict in Land-Use Planning Using a Values-Centered E-Participation Tool: A Canadian Case Study in Aggregate Mining. In *Proceedings of the 52nd Hawaii International Conference on System Sciences*. doi:10.24251/hicss.2019.410

Pilgrim, E. S., Macleod, C. J. A., Blackwell, M. S. A., Bol, R., Hogan, D. V., Chadwick, D. R., Cardenas, L., Misselbrook, T. H., Haygarth, P. M., Brazier, R. E., Hobbs, P., Hodgson, C., Jarvis, S., Dungait, J., Murray, P. J., and Firbank, L. G. (2010). Interactions among agricultural production and other ecosystem services delivered from european temperate grassland systems. *Advances in Agronomy*, **109**, 117–154.

Pittock, J. (2011). National climate change policies and sustainable water management: Conflicts and synergies. *Ecology and Society*, **16**(2). doi:25

Plieninger, T., Torralba, M., Hartel, T., and Fagerholm, N. (2019). Perceived ecosystem services synergies, trade-offs, and bundles in European high nature value farming landscapes. *Landscape Ecology*. doi:10.1007/s10980-019-00775-1

Pocewicz, A., Nielsen-Pincus, M., Brown, G., and Schnitzer, R. (2012). An Evaluation of Internet Versus Paper-based Methods for Public Participation Geographic Information Systems (PPGIS). *Transactions in GIS*, **16**(1), 39–53.

Porras, G. L., Stringer, L. C., and Quinn, C. H. (2018). Unravelling Stakeholder Perceptions to Enable AdaptiveWater Governance in Dryland Systems. *Water Resources Management*, 1–17.

Prell, C., Hubacek, K., Reed, M., Quinn, C., Jin, N., Holden, J., Burt, T., Kirby, M., and Sendzimir, J. (2007). If you have a hammer everything looks like a nail: traditional versus participatory model building. *Interdisciplinary Science Reviews*, **32**(3), 20.

Psaltopoulos, D., Wade, A. J., Skuras, D., Kernan, M., Tyllianakis, E., and Erlandsson, M. (2017). False positive and false negative errors in the design and implementation of agri-environmental policies: A case study on water quality and agricultural nutrients. *Science of the Total Environment*, **575**. doi:10.1016/j.scitotenv.2016.09.181

Puma, M. J. (2019). Resilience of the global food system. *Nature Sustainability*, pp. 260–261.

Punjabi, B., and Johnson, C. A. (2019). The politics of rural–urban water conflict in India: Untapping the power of institutional reform. *World Development*, **120**, 182–192.

Quilliam, R. S., Kinzelman, J., Brunner, J., and Oliver, D. M. (2015). Resolving conflicts in public health protection and ecosystem service provision at designated bathing waters. *Journal of Environmental Management*, pp. 237–242.

Randall, N. P., Donnison, L. M., Lewis, P. J., and James, K. L. (2015). How effective are on-farm mitigation measures for delivering an improved water environment? A systematic map. *Environmental Evidence*, **4**(1). doi:10.1186/s13750-015-0044-5

Raudsepp-Hearne, C., Peterson, G. D., and Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, **107**(11), 5242–5247.

Raum, S. (2018). A framework for integrating systematic stakeholder analysis in ecosystem services research: Stakeholder mapping for forest ecosystem services in the UK. *Ecosystem Services*, **29**, 170–184.

Raymond, C. M., Bieling, C., Fagerholm, N., Martin-Lopez, B., and Plieninger, T. (2016). The farmer as a landscape steward: Comparing local understandings of landscape stewardship, landscape values, and land management actions. *Ambio*. doi:10.1007/s13280-015-0694-0

Reaney, S. M., Mackay, E. B., Haygarth, P. M., Fisher, M., Molineux, A., Potts, M., and Benskin, C. M. W. H. (2019). Identifying critical source areas using multiple methods for effective diffuse pollution mitigation. *Journal of Environmental Management*, **250**. doi:10.1016/j.jenvman.2019.109366

Redman, S., Greenhalgh, T., Adedokun, L., Staniszewska, S., and Denegri, S. (2021). Co-production of knowledge: The future. *The BMJ*. doi:10.1136/bmj.n434

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation*, pp. 2417–2431.

Reed, M. S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., Quinn, C. H., and Stringer, L. C. (2009). Who's in and why? A typology of stakeholder analysis methods for natural resource management. *Journal of Environmental Management*. doi:10.1016/j.jenvman.2009.01.001

Reed, M. S., Moxey, A., Prager, K., Hanley, N., Skates, J., Bonn, A., Evans, C. D., Glenk, K., and Thomson, K. (2014). Improving the link between payments and the provision of ecosystem services in agri-environment schemes. *Ecosystem Services*. doi:10.1016/j.ecoser.2014.06.008

Reed, M. S., Vella, S., Challies, E., de Vente, J., Frewer, L., Hohenwallner-Ries, D., Huber, T., Neumann, R. K., Oughton, E. A., Sidoli del Ceno, J., and van Delden, H. (2018). A theory of participation: what makes stakeholder and public engagement in environmental management work? *Restoration Ecology*, pp. S7–S17.

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T. J., Kidd, K. A., MacCormack, T. J., Olden, J. D., Ormerod, S. J., Smol, J. P., Taylor, W. W., Tockner, K., Vermaire, J. C., Dudgeon, D., and Cooke, S. J. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews*, **94**(3), 849–873.

Reilly, K., Adamowski, J., and John, K. (2018). Participatory mapping of ecosystem services to understand stakeholders' perceptions of the future of the Mactaquac Dam, Canada. *Ecosystem Services*, **30**, 107–123.

Rimmert, M., Baudoin, L., Cotta, B., Kochskämper, E., and Newig, J. (2020). Participation in river basin planning under the water framework directive-Has it benefitted good water status? *Water Alternatives*, **13**(3), 484–512.

Rittel, H. W. J., and Webber, M. M. (1973). Dilemmas in a General Theory of Planning. *Policy Sciences*, **4**, 155–169.

Robbins, P. (2004). Political ecology: A critical introduction. In *Critical Introductions to Geography*, pp. 3–16.

Rodríguez, J. P., Beard, T. D. J., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., Dobson, A. P., and Peterson, G. D. (2006). Trade-offs across Space, Time, and Ecosystem Services. *Ecology and Society*, **11**(1), 28.

Rogers, B. C., Bertram, N., Gersonius, B., Gunn, A., Löwe, R., Murphy, C., Pasman, R., Radhakrishnan, M., Urich, C., Wong, T. H. F., and Arnbjerg-Nielsen, K. (2020). An interdisciplinary and catchment approach to enhancing urban flood resilience: A Melbourne case. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, **378**(2168). doi:10.1098/rsta.2019.0201

Rollason, E., Bracken, L. J., Hardy, R. J., and Large, A. R. G. (2018). Evaluating the success of public participation in integrated catchment management. *Journal of Environmental Management*, **228**, 267–278.

Roque de Oliveira, A., and Partidário, M. (2020). You see what I mean? – A review of visual tools for inclusive public participation in EIA decision-making processes. *Environmental Impact Assessment Review*. doi:10.1016/j.eiar.2020.106413

Rose, N. A., and Parsons, E. C. M. (2015). "Back off, man, I'm a scientist!" When marine conservation science meets policy. *Ocean and Coastal Management*, **115**, 71–76.

Rouillard, J. J., and Spray, C. J. (2017). Working across scales in integrated catchment management: lessons learned for adaptive water governance from regional experiences. *Regional Environmental Change*, **17**(7). doi:10.1007/s10113-016-0988-1

RStudio. (2016). RStudio: Integrated development for R. *[Online] RStudio, Inc., Boston, MA URL Http://Www. Rstudio. Com*. doi:10.1007/978-81-322-2340-5

Rust, W., and Venn, P. (2018). Are the benefits of integrated catchment modelling being realized in the united kingdom? *International Journal of Environmental Impacts: Management, Mitigation and Recovery*, **1**(3). doi:10.2495/ei-v1-n3-232-239

Saad-Sulonen, J., Eriksson, E., Halskov, K., Karasti, H., and Vines, J. (2018). Unfolding participation over time: temporal lenses in participatory design. *CoDesign*, **14**(1), 4–16.

Sachs, J. D., Schmidt-Traub, G., Mazzucato, M., Messner, D., Nakicenovic, N., and Rockström, J. (2019). Six Transformations to achieve the Sustainable Development Goals. *Nature Sustainability*, **2**(9), 805–814.

Sahle, M., Saito, O., Fürst, C., and Yeshitela, K. (2019). Quantifying and mapping of water-related ecosystem services for enhancing the security of the food-water-energy nexus in tropical data–sparse catchment. *Science of the Total Environment*, **646**. doi:10.1016/j.scitotenv.2018.07.347

Satz, D., Gould, R. K., Chan, K. M. A., Guerry, A., Norton, B., Satterfield, T., Halpern, B. S., Levine, J., Woodside, U., Hannahs, N., Basurto, X., and Klain, S. (2013). The challenges of incorporating cultural ecosystem services into environmental assessment. *Ambio*, pp. 675–684.

Saunders, F. P. (2016). Complex Shades of Green: Gradually Changing Notions of the 'Good Farmer' in a Swedish Context. *Sociologia Ruralis*. doi:10.1111/soru.12115

Sayles, J. S., and Baggio, J. A. (2017). Social-ecological network analysis of scale mismatches in estuary watershed restoration. *Proceedings of the National Academy of Sciences of the United States of America*, **114**(10), E1776–E1785.

Sayles, J. S., Mancilla Garcia, M., Hamilton, M., Alexander, S. M., Baggio, J. A., Fischer, A. P., Ingold, K., Meredith, G. R., and Pittman, J. (2019). Social-ecological network analysis for sustainability sciences: A systematic review and innovative research agenda for the future. *Environmental Research Letters*. doi:10.1088/1748-9326/ab2619

Schlosser, C. A., Strzepek, K., Gao, X., Fant, C., Blanc, É., Paltsev, S., Jacoby, H., Reilly, J., and Gueneau, A. (2014). The future of global water stress: An integrated assessment. *Earth's Future*, **2**(8), 341–361.

Schmeller, D. S., Courchamp, F., and Killeen, G. (2020). Biodiversity loss, emerging pathogens and human health risks. *Biodiversity and Conservation*, pp. 3095–3102.

Schomers, S., and Matzdorf, B. (2013). Payments for ecosystem services: A review and comparison of developing and industrialized countries. *Ecosystem Services*, **6**, 16–30.

Schröder, N. J. S., Newig, J., and Watson, N. (2020). Bright spots for local WFD implementation through collaboration with nature conservation authorities? *Water Alternatives*, **13**(3).

Schröter, B., Matzdorf, B., Sattler, C., and Garcia Alarcon, G. (2015). Intermediaries to foster the implementation of innovative land management practice for ecosystem service provision - A new role for researchers. *Ecosystem Services*, **16**, 192–200.

Schulp, C. J. E., Burkhard, B., Maes, J., Vliet, J. Van, and Verburg, P. H. (2014). Uncertainties in Ecosystem Service Maps : A Comparison on the European Scale, **9**(10). doi:10.1371/journal.pone.0109643

Schwarz, H., Revilla, M., and Weber, W. (2020). Memory effects in repeated survey questions reviving the empirical investigationassumptionof the independent measurements assumption. *Survey Research Methods*, **14**(3). doi:10.18148/srm/2020.v14i3.7579

Scottish Government, T. (2016). *Key Scottish Environment: Statistics 2016 - October 2016*. Retrieved from http://www.gov.scot/Topics/Statistics/Browse/Environment/Publications

Semeraro, T., Giannuzzi, C., Beccarisi, L., Aretano, R., De Marco, A., Pasimeni, M. R., Zurlini, G., and Petrosillo, I. (2015). A constructed treatment wetland as an opportunity to enhance biodiversity and ecosystem services. *Ecological Engineering*, **82**, 517–526.

Seppelt, R., Dormann, C. F., Eppink, F. V, Lautenbach, S., and Schmidt, S. (2011). A quantitative review of ecosystem service studies : approaches , shortcomings and the road ahead, 630–636.

Seppelt, R., Fath, B., Burkhard, B., Fisher, J. L., Grêt-Regamey, A., Lautenbach, S., Pert, P., Hotes, S., Spangenberg, J., Verburg, P. H., and Van Oudenhoven, A. P. E. (2012). Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecological Indicators*, **21**, 145–154.

Shepherd, E., Milner-Gulland, E. J., Knight, A. T., Ling, M. A., Darrah, S., van Soesbergen, A., and Burgess, N. D. (2016). Status and Trends in Global Ecosystem Services and Natural Capital: Assessing Progress Toward Aichi Biodiversity Target 14. *Conservation Letters*, pp. 429–437.

Shi, H., Luo, G., Zheng, H., Chen, C., Bai, J., Liu, T., Ochege, F. U., and De Maeyer, P. (2020). Coupling the water-energy-food-ecology nexus into a Bayesian network for water resources analysis and management in the Syr Darya River basin. *Journal of Hydrology*, **581**. doi:10.1016/j.jhydrol.2019.124387

Shoji, Y., Kim, H., Kubo, T., Tsuge, T., Aikoh, T., and Kuriyama, K. (2021). Understanding preferences for pricing policies in Japan's national parks using the best–worst scaling method. *Journal for Nature Conservation*, **60**. doi:10.1016/j.jnc.2021.125954

Sing, L., Metzger, M. J., Paterson, J. S., and Ray, D. (2018). A review of the effects of forest management intensity on ecosystem services for northern European temperate forests with a focus on the UK. *Forestry*. doi:10.1093/forestry/cpx042

Singh, G. G., Sinner, J., Ellis, J., Kandlikar, M., Halpern, B. S., Satterfield, T., and Chan, K. M. A. (2017).

Mechanisms and risk of cumulative impacts to coastal ecosystem services: An expert elicitation approach. *Journal of Environmental Management*, **199**, 229–241.

Singletary, L., and Sterle, K. (2020). Supporting local adaptation through the co-production of climate information: An evaluation of collaborative research processes and outcomes. *Climate Services*, **20**. doi:10.1016/j.cliser.2020.100201

Skaalsveen, K., Ingram, J., and Urquhart, J. (2020). The role of farmers' social networks in the implementation of no-till farming practices. *Agricultural Systems*, **181**. doi:10.1016/j.agsy.2020.102824

Skarlatidou, A., Suskevics, M., Göbel, C., Prūse, B., Tauginiené, L., Mascarenhas, A., Mazzonetto, M., Sheppard, A., Barrett, J., Haklay, M., Baruch, A., Moraitopoulou, E.-A., Austen, K., Baïz, I., Berditchevskaia, A., Berényi, E., Hoyte, S., … Wyszomirski, P. (2019). The Value of Stakeholder Mapping to Enhance Co-Creation in Citizen Science Initiatives. *Citizen Science: Theory and Practice*. doi:10.5334/cstp.226

Smith, L. E. D., and Porter, K. S. (2010). Management of catchments for the protection of water resources: Drawing on the New York City watershed experience. *Regional Environmental Change*, **10**(4), 311–326.

Smith, P. (2013). Delivering food security without increasing pressure on land. *Global Food Security*, pp. 18–23.

Smith, R. I., Barton, D. N., Dick, J., Haines-Young, R., Madsen, A. L., Rusch, G. M., Termansen, M., Woods, H., Carvalho, L., Giucă, R. C., Luque, S., Odee, D., Rusch, V., Saarikoski, H., Adamescu, C. M., Dunford, R., Ochieng, J., … Vikström, S. (2018). Operationalising ecosystem service assessment in Bayesian Belief Networks: Experiences within the OpenNESS project. *Ecosystem Services*, **29**. doi:10.1016/j.ecoser.2017.11.004

Software, S. (2007). *The MaxDiff/Web Technical Paper, v6.0. Sawtooth Software, Inc., Sequim, Washington.*

Soto, J. R., Escobedo, F. J., Khachatryan, H., and Adams, D. C. (2018). Consumer demand for urban forest ecosystem services and disservices: Examining trade-offs using choice experiments and best-worst scaling. *Ecosystem Services*, **29**. doi:10.1016/j.ecoser.2017.11.009

South Esk Catchment Partnership Steering Group. (2009). *The South Esk CatchmentManagement Plan*.

Sova, C. A., Thornton, T. F., Zougmore, R., Helfgott, A., and Chaudhury, A. S. (2016). Power and influence mapping in Ghana's agricultural adaptation policy regime. *Climate and Development*, **5529**(April), 1–16.

Spake, R., Lasseur, R., Crouzat, E., Bullock, J. M., Lavorel, S., Parks, K. E., Schaafsma, M., Bennett, E. M., Maes, J., Mulligan, M., Mouchet, M., Peterson, G. D., Schulp, C. J. E., Thuiller, W., Turner, M. G., Verburg, P. H., and Eigenbrod, F. (2017). Unpacking ecosystem service bundles: Towards predictive mapping of synergies and trade-offs between ecosystem services. *Global Environmental Change*, **47**, 37–50.

Spash, C. L. (2012). New foundations for ecological economics. *Ecological Economics*, **77**, 36–47.

Spears, B. M., Carvalho, L., Perkins, R., Kirika, a., and Paterson, D. M. (2012). Long-term variation and regulation of internal phosphorus loading in Loch Leven. *Hydrobiologia*, **681**, 23–33.

Stepanova, O. (2015). Conflict resolution in coastal resource management: Comparative analysis of case studies from four European countries. *Ocean and Coastal Management*, **103**, 109–122.

Sterling, S. M., Ducharne, A., and Polcher, J. (2012). The impact of global land-cover change on the terrestrial water cycle. *Nature Climate Change*, **2**(10), 1–6.

Stevens, C. J., Quinton, J. N., and Systems, A. (2009). Diffuse Pollution Swapping in Arable Agricultural Systems. *Critical Reviews in Environmental Science and Technology*, **39**(6), 478–520.

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., de Snoo, G. R., Rakosy, L., and Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe - A review. *Journal of Environmental Management*, pp. 22–46.

Stoeckl, N., Chaiechi, T., Farr, M., Jarvis, D., ??lvarez-Romero, J. G., Kennard, M. J., Hermoso, V., and Pressey, R. L. (2015). Co-benefits and trade-offs between agriculture and conservation: A case study in Northern Australia. *Biological Conservation*, **191**, 478–494.

Stosch, K. C., Quilliam, R. S., Bunnefeld, N., and Oliver, D. M. (2019). Quantifying stakeholder understanding of an ecosystem service trade-off. *Science of the Total Environment*, **651**, 2524–2534.

Stosch, K., Quilliam, R., Bunnefeld, N., and Oliver, D. (2017). Managing Multiple Catchment Demands for Sustainable Water Use and Ecosystem Service Provision. *Water*, **9**(9), 677.

Strokal, M., Ma, L., Bai, Z., Luan, S., Kroeze, C., Oenema, O., Velthof, G., and Zhang, F. (2016). Alarming nutrient pollution of Chinese rivers as a result of agricultural transitions. *Environmental Research Letters*, **11**(2). doi:10.1088/1748-9326/11/2/024014

Stutter, M., Kronvang, B., Ó hUallacháin, D., and Rozemeijer, J. (2019). Current Insights into the Effectiveness of Riparian Management, Attainment of Multiple Benefits, and Potential Technical Enhancements. *Journal of Environmental Quality*, **48**(2). doi:10.2134/jeq2019.01.0020

Thaler, T., and Levin-Keitel, M. (2016). Multi-level stakeholder engagement in flood risk management-A question of roles and power: Lessons from England. *Environmental Science and Policy*, **55**. doi:10.1016/j.envsci.2015.04.007

Thomas, E., Riley, M., and Spees, J. (2019). Good farming beyond farmland – Riparian environments and the concept of the 'good farmer.' *Journal of Rural Studies*, **67**. doi:10.1016/j.jrurstud.2019.02.015

Trabucchi, M., Ntshotsho, P., Farrell, P. O., and Comín, F. A. (2012). Ecosystem service trends in basin-scale restoration initiatives : A review. *Journal of Environmental Management*, **111**, 18–23.

Tromp-Van Meerveld, H. J., and McDonnell, J. J. (2006). Threshold relations in subsurface stormflow: 1. A 147-storm analysis of the Panola hillslope. *Water Resources Research*, **42**(2). doi:10.1029/2004WR003778

Tulloch, V. J. D., Tulloch, A. I. T., Visconti, P., Halpern, B. S., Watson, J. E. M., Evans, M. C., Auerbach, N. a, Barnes, M., Beger, M., Chadès, I., Giakoumi, S., McDonald-Madden, E., Murray, N. J., Ringma, J., and Possingham, H. P. (2015). Why do we map threats? Linking threat mapping with actions to make better conservation decisions. *Frontiers in Ecology and the Environment*, **13**(2), 91–99.

Turkelboom, F., Leone, M., Jacobs, S., Kelemen, E., García-Llorente, M., Baró, F., Termansen, M., Barton, D. N., Berry, P., Stange, E., Thoonen, M., Kalóczkai, Á., Vadineanu, A., Castro, A. J., Czúcz, B., Röckmann, C., Wurbs, D., … Rusch, V. (2018). When we cannot have it all: Ecosystem services trade-offs in the context of spatial planning. *Ecosystem Services*, **29**, 566–578.

Turner, K. G., Anderson, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., Ratna, N., Sandhu, H., Sutton, P. C., Svenning, J.-C., Turner, G. M., Varennes, Y.-D., Voinov, A., … Turner, K. G. (2016). A review of methods , data , and models to assess changes in the value of ecosystem services from land degradation and restoration. *Ecological Modelling*, **319**, 190–207.

Turner, K. G., Odgaard, M. V., Bøcher, P. K., Dalgaard, T., and Svenning, J. C. (2014). Bundling ecosystem services in Denmark: Trade-offs and synergies in a cultural landscape. *Landscape and Urban Planning*, **125**, 89–104.

Tyner, E. H., and Boyer, T. A. (2020). Applying best-worst scaling to rank ecosystem and economic benefits of restoration and conservation in the Great Lakes. *Journal of Environmental Management*, **255**. doi:10.1016/j.jenvman.2019.109888

UNESCO. (2019). *The United Nations World Water Development Report 2019: Leaving no one behind. UNESCO Digital Library*.

United Nations Development Programme. (2016). Human Development Report 2016: Human Development for Everyone. *United Nations Development Programme*, 286.

Urbanek, S. (2003). tiff: Read and write TIFF images. R package version 0.1-5.

Uthes, S., and Matzdorf, B. (2013). Studies on agri-environmental measures: A survey of the literature. *Environmental Management*, **51**(1), 251–266.

Vallet, A., Locatelli, B., Levrel, H., Wunder, S., Seppelt, R., Scholes, R. J., and Oszwald, J. (2018). Relationships Between Ecosystem Services: Comparing Methods for Assessing Tradeoffs and Synergies. *Ecological Economics*, **150**, 96–106.

van Delden, H., Seppelt, R., White, R., and Jakeman, a. J. (2011). A methodology for the design and development of integrated models for policy support. *Environmental Modelling & Software*, **26**(3), 266–279.

van den Heuvel, L., Blicharska, M., Masia, S., Sušnik, J., and Teutschbein, C. (2020). Ecosystem services in the Swedish water-energy-food-land-climate nexus: Anthropogenic pressures and physical interactions. *Ecosystem Services*, **44**. doi:10.1016/j.ecoser.2020.101141

van Herzele, A., Gobin, A., Van Gossum, P., Acosta, L., Waas, T., Dendoncker, N., and Henry de Frahan, B. (2013). Effort for money? Farmers' rationale for participation in agri-environment measures with different implementation complexity. *Journal of Environmental Management*, **131**. doi:10.1016/j.jenvman.2013.09.030

Vasslides, J. M., and Jensen, O. P. (2016). Fuzzy cognitive mapping in support of integrated ecosystem assessments: Developing a shared conceptual model among stakeholders. *Journal of Environmental Management*, **166**, 348–356.

Verhagen, W., van der Zanden, E. H., Strauch, M., van Teeffelen, A. J. A., and Verburg, P. H. (2018). Optimizing the allocation of agri-environment measures to navigate the trade-offs between ecosystem services, biodiversity and agricultural production. *Environmental Science and Policy*, **84**. doi:10.1016/j.envsci.2018.03.013

Vermaat, J. E., Wagtendonk, A. J., Brouwer, R., Sheremet, O., Ansink, E., Brockhoff, T., Plug, M., Hellsten, S., Aroviita, J., Tylec, L., Giełczewski, M., Kohut, L., Brabec, K., Haverkamp, J., Poppe, M., Böck, K., Coerssen, M., … Hering, D. (2016). Assessing the societal benefits of river restoration using the ecosystem services approach. *Hydrobiologia*, **769**(1). doi:10.1007/s10750-015-2482-z

Vesterager, J. P., Teilmann, K., and Vejre, H. (2012). Assessing long-term sustainable environmental impacts of agri-environment schemes on land use. *European Journal of Forest Research*, **131**(1). doi:10.1007/s10342-010-0469-x

Villa, F., Bagstad, K. J., Voigt, B., Johnson, G. W., Portela, R., Honzák, M., and Batker, D. (2014). A methodology for adaptable and robust ecosystem services assessment. *PLoS ONE*, **9**(3). doi:10.1371/journal.pone.0091001

Vito, L. De, Fairbrother, M., and Russel, D. (2020). Implementing the water framework directive and tackling diffuse pollution from agriculture: Lessons from England and Scotland. *Water (Switzerland)*, **12**(1). doi:10.3390/w12010244

Von Der Dunk, A., Grêt-Regamey, A., Dalang, T., and Hersperger, A. M. (2011). Defining a typology of peri-urban land-use conflicts - A case study from Switzerland. *Landscape and Urban Planning*, **101**(2), 149–156.

von Stackelberg, K. E. (2013). Decision analytic strategies for integrating ecosystem services and risk assessment. *Integrated Environmental Assessment and Management*, **9**(2), 260–268.

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, a, Green, P., Glidden, S., Bunn, S. E., Sullivan, C. a, Liermann, C. R., and Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, **467**(7315), 555–561.

Walker, W. E., Harremoës, P., Rotmans, J., van der Sluijs, J. P., van Asselt, M. B. a., Janssen, P., and Krayer von Krauss, M. P. (2003). Defining Uncertainty: A Conceptual Basis for Uncertainty Management in Model-Based Decision Support. *Integrated Assessment*, **4**(1), 5–17.

Wam, H. K., Bunnefeld, N., Clarke, N., and Hofstad, O. (2016). Conflicting interests of ecosystem services: multi-criteria modelling and indirect evaluation of trade-offs between monetary and non-monetary measures. *Ecosystem Services*, **in press**.

Ward, M., Possingham, H., Rhodes, J. R., and Mumby, P. (2018). Food, money and lobsters: Valuing ecosystem services to align environmental management with Sustainable Development Goals. *Ecosystem Services*, **29**, 56–69.

Warner, J. (2007). The Beauty of the Beast : Multi-Stakeholder Participation for Integrated Catchment Management. *Multi-Stakeholder Platforms for Integrated Water Management*.

Watson, N., Shrubsole, D., and Mitchell, B. (2019). Governance arrangements for integrated water resources management in Ontario, Canada, and Oregon, USA: Evolution and lessons. *Water (Switzerland)*, **11**(4). doi:10.3390/w11040663

Waylen, K. A., Blackstock, K. L., Marshall, K. B., and Dunglinson, J. (2015). Participation–Prescription Tension in Natural Resource Management: The case of diffuse pollution in Scottish water management. *Environmental Policy and Governance*, **25**(2). doi:10.1002/eet.1666

Weijerman, M., Gove, J. M., Williams, I. D., Walsh, W. J., Minton, D., and Polovina, J. J. (2018). Evaluating management strategies to optimise coral reef ecosystem services. *Journal of Applied Ecology*. doi:10.1111/1365-2664.13105

Westerink, J., Opdam, P., van Rooij, S., and Steingröver, E. (2017). Landscape services as boundary concept in landscape governance: Building social capital in collaboration and adapting the landscape. *Land Use Policy*. doi:10.1016/j.landusepol.2016.11.006

Westphal, C., Vidal, S., Horgan, F. G., Gurr, G. M., Escalada, M., Van Chien, H., Tscharntke, T., Heong, K. L., and Settele, J. (2015). Promoting multiple ecosystem services with flower strips and participatory approaches in rice production landscapes. *Basic and Applied Ecology*, **16**(8), 681–

689.

Whitman, G. P., Pain, R., and Milledge, D. G. (2015). Going with the flow ? Using participatory action research in physical geography. *Progress in Physical Geography*, 1–18.

Whittingham, M. J. (2007). Will agri-environment schemes deliver substantial biodiversity gain, and if not why not? *Journal of Applied Ecology*. doi:10.1111/j.1365-2664.2006.01263.x

Wilkes-Allemann, J., Pütz, M., Hirschi, C., and Fischer, C. (2015). Conflict situations and response strategies in urban forests in Switzerland. *Scandinavian Journal of Forest Research*, **30**(3), 204–216.

Withey, J. C., Lawler, J. J., Polasky, S., Plantinga, A. J., Nelson, E. J., Kareiva, P., Wilsey, C. B., Schloss, C. A., Nogeire, T. M., Ruesch, A., Ramos, J., and Reid, W. (2012). Maximising return on conservation investment in the conterminous USA. *Ecology Letters*, **15**(11), 1249–1256.

Wunder, S., Engel, S., and Pagiola, S. (2008). Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics*, **65**(4), 834–852.

Yang, L. E., Chan, F. K. S., and Scheffran, J. (2018). Climate change, water management and stakeholder analysis in the Dongjiang River basin in South China. *International Journal of Water Resources Development*, **34**(2), 166–191.

Ye, B., Zhang, X., Zhang, X., and Zheng, C. (2020). Climate change, environmental impact, and human health. *Environmental Geochemistry and Health*, pp. 715–717.

Young, J. C., Thompson, D. B. A., Moore, P., MacGugan, A., Watt, A., and Redpath, S. M. (2016). A conflict management tool for conservation agencies. *Journal of Applied Ecology*, **53**(3), 705–711.

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Henle, K., Johnson, R., Laczko, E., McCracken, D., Matouch, S., Niemela, J., and Richards, C. (2005). Towards sustainable land use: Identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodiversity and Conservation*, pp. 1641–1661.

Zhang, Y., Collins, A. L., Jones, J. I., Johnes, P. J., Inman, A., and Freer, J. E. (2017). The potential benefits of on-farm mitigation scenarios for reducing multiple pollutant loadings in prioritised agri-environment areas across England. *Environmental Science and Policy*, **73**. doi:10.1016/j.envsci.2017.04.004

Zheng, H., Li, Y., Robinson, B. E., Liu, G., Ma, D., Wang, F., Lu, F., Ouyang, Z., and Daily, G. C. (2016). Using ecosystem service trade-offs to inform water conservation policies and management practices. *Frontiers in Ecology and the Environment*, **14**(10), 527–532.

Zingraff-Hamed, A., Hüesker, F., Lupp, G., Begg, C., Huang, J., Oen, A., Vojinovic, Z., Kuhlicke, C., and Pauleit, S. (2020). Stakeholder mapping to co-create nature-based solutions: Who is on board? *Sustainability (Switzerland)*. doi:10.3390/su12208625

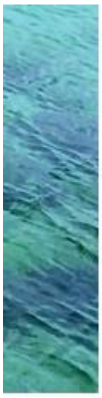
Zoderer, B. M., Tasser, E., Carver, S., and Tappeiner, U. (2019). An integrated method for the mapping of landscape preferences at the regional scale. *Ecological Indicators*, **106**. doi:10.1016/j.ecolind.2019.05.061

Zolkafli, A., Liu, Y., and Brown, G. (2017). Bridging the knowledge divide between public and experts using PGIS for land use planning in Malaysia. *Applied Geography*, **83**, 107–117.

Appendix 1: Interview protocol and questionnaire Chapter 3.

The purpose of the study was explained to participants, along with confidentiality procedures on paper which was signed to express their consent. Participants were then shown a map of their catchments that included land cover data and were given time to familiarise themselves with the legend.
Q1: *Looking at this map, what are the three most significant ecosystem services the catchment provides?*
Q2: *Can these services be supplied unimpeded by one another from the catchment or do they compete for space?*
Participants were then shown a map of their catchments that included land cover data and in which the catchment was split by two lines into equal areas into an upstream, middle, and downstream area.
Q3: *Looking at the map, what are the three most significant ecosystem services each zone provides?*
Q4: *Within the zones, do these three services compete for space? Can you rank the three zones from lowest to highest competition between land uses?*
Q5: *Are there pressures from stakeholder groups to increase or reduce any of these ecosystem service provisions?*
Q6: *If so, has this caused conflict with other stakeholder groups?*
Q7: *if not, are there any other ecosystem services or land uses which stakeholder groups want to increase or reduce and has this caused conflict between groups?*
Participants were now given a permanent marker pen and asked to draw and annotate as follows.
Q8: *Which areas in the catchment do you believe to be under most conflict or pressure from competing land use and why? You may identify these by drawing on the map, and/or selecting a land cover type.*
Q9: *What are the major drivers for conflict between land use, ecosystem service provision and stakeholders in your catchment?*
Q10: *What do you think could be done to reduce conflict between stakeholders in your catchment?*
Q11: *By 2030, what will be the major drivers for conflict between land use, ecosystem service provision and stakeholders in your catchment?*
Q12: *When the United Kingdom leaves the European Union, do you expect land use, ecosystem service provision and stakeholder conflict in your catchment to change?*

Appendix 2: Flyer for potential participants for Chapter 5.



Are we getting multiple benefits from agri-environment payments?

A Scottish multi-stakeholder survey

Please contribute **20 minutes** of your time to help us find out which measures are preferable for farming, water quality, flooding and biodiversity

About the study:

Take part in this quick online survey to help us capture the knowledge and understanding of a range of relevant stakeholder groups on:

- **Which agri-environment schemes have the best and worst outcomes in terms of delivering a number of environmental improvements?**
- **What are the main barriers to maximising benefits of agri-environment schemes in Scotland?**

If you would like to contribute your views please follow the link to the survey:

https://AESBWS.sawtoothsoftware.com/login.html

About the project:

I am Kathleen Stosch, currently working on my PhD thesis at the University of Stirling. This study will form part of my work on how we can balance the often opposing demands of different stakeholder groups and promote greater cooperation and more integrated decision-making.

Scottish Government
gov.scot
Hydro Nation Scholars Programme

Any questions about my PhD work or this study, please do not hesitate to contact me: **kathleen.stosch@stir.ac.uk**

UNIVERSITY *of* STIRLING

Appendix 3: Online survey questions for Chapter 5.

Participant information:

Stakeholder prioritisation of agri-environment schemes and their multiple environmental benefits

This survey should take around 20 minutes and is designed to capture your knowledge on a number of agri-environment schemes that are aimed to reduce diffuse pollution, downstream flooding and improve biodiversity. Your responses will provide insight into the wider understanding of some of the benefits and challenges of land and water management in Scotland.

The study forms part of my PhD studentship at the University of Stirling, which is funded by a Scottish Government Hydro Nation Scholarship. The aim of my research is to devise strategies to promote collaboration as opposed to conflict in managing ecosystem services in river basins and find ways to optimise landscape-scale ecosystem service delivery in catchments.

If you have any questions about my research, please feel free to contact me (Kathleen Stosch) via phone at 07766800934 or email at kathleen.stosch@stir.ac.uk

Thank you for participating.

Click the 'Next' button to continue...

Consent Form:
Your responses to the survey will be recorded on an anonymous online platform. The data will be accessed, transcribed and analysed only by the interviewer (Kathleen Stosch) and will be stored only on their password protected personal computer. We will collect no personal information from you. Your contact details will be kept private and confidential. Identifiable information will not be used in any publication or presentation. We will not pass your details on to any organisation or company. Individual responses to the survey will be collated so that stakeholder group-wide generalisations can be inferred.

Please tick the boxes to indicate that:

- ☐ You confirm that you have read and understand the participant information explaining the research and have had the opportunity to ask questions about the project.
- ☐ You understand that your participation is voluntary and that you are free to withdraw at any time and withdraw your data within two weeks without giving reason. You may decline to answer any particular question, without giving reason.
- ☐ You agree to take part in the study.

Click the 'Next' button to continue...

Please state what stakeholder group you belong to:

- ☐ Environmental regulator
- ☐ Farmer
- ☐ Farm advisor
- ☐ Fisheries trust
- ☐ Local Authority
- ☐ National park authority
- ☐ Scientist
- ☐ Water regulator
- ☐ Other […]

What is your job role?
[...]

Please estimate how many years you have worked in that (or a similar) role:

- Less than one year
- 1-2 years
- 2-5 years
- 5-10 years
- more than 10 years
- Don't know/not sure

Please name 1-3 Scottish river catchments that you have experience working in.
[...]

Click the 'Next' button to continue...

The following questions are designed to capture your knowledge on the benefits of different agri-environment schemes for reducing diffuse pollution, downstream flooding and improving biodiversity.

Which of the following areas do you have the greatest expertise on?

- Biodiversity impacts of land management
- Diffuse pollution impacts of land management
- Flooding impacts of land management

Click the 'Next' button to continue...

The environmental benefits of agri-environment schemes
As catchments are highly variable in geology, topography, land cover and land management, please do not answer the following questions based on a specific catchment you have in mind.

Instead, broadly consider the wider benefits of the agri-environment measures if they were implemented across farms in a model Scottish catchment, with a mixture of typical, current farming practices. Agricultural land use in this catchment consists of upland rough grazing (30% of land cover), lowland improved grazing (15% land cover) as well as arable agriculture common in Scotland such as growing barley, potatoes and oilseed rape (10% of land cover).

Please estimate the likely environmental benefits of the measures, assuming they were utilised widely in the catchment (in around 50% of all farms) and that placement was in conjunction with best farm advice to gain maximum benefits from the measure.

Click the 'Next' button to continue...

Your perception on the ability of agri-environment schemes to <u>decrease diffuse pollution</u>.

Please go through the following ten choice exercises and select the measures you estimate to be the best and the worst at <u>reducing diffuse pollution</u> in the catchment.

Click the 'Next' button to continue...

Reducing Diffuse Pollution

Please picture a model Scottish catchment with a land cover of 30% rough grazing, 15% improved grazing and 10% arable agriculture.

Considering only the following 4 measures, which do you think is most likely and which is least likely to reduce diffuse pollution from farm land to downstream water courses, assuming they were utilised widely in the catchment (in around 50% of all farms)?

[Ten sets of BWS questions across ten pages with above heading]

Click the 'Next' button to continue...

Your perception on the ability of agri-environment schemes to increase biodiversity.

Please go through the following ten choice exercises and select the measures you estimate to be the best and the worst at improving biodiversity in the catchment.

Click the 'Next' button to continue...

Supporting Biodiversity

Please picture a model Scottish catchment with a land cover of 30% rough grazing, 15% improved grazing and 10% arable agriculture.

Considering only the following 4 measures, which do you think is most likely and which is least likely to support biodiversity on farm land, assuming they were utilised widely in the catchment (in around 50% of all farms)?

[Ten sets of BWS questions across ten pages with above heading]

Click the 'Next' button to continue...

Your perception on the ability of agri-environment schemes to improve flood alleviation.

Please go through the following ten choice exercises and select the measures you estimate to be the best and the worst at alleviating flooding in the catchment.

Click the 'Next' button to continue...

Alleviating Downstream Flooding

Please picture a model Scottish catchment with a land cover of 30% rough grazing, 15% improved grazing and 10% arable agriculture.

Considering only the following 4 measures, which do you think is most likely and which is least likely to alleviate downstream flooding, assuming they were utilised widely in the catchment (in around 50% of all farms)?

[Ten sets of BWS questions across ten pages with above heading]

Click the 'Next' button to continue...

How common do you believe the current uptake of these agri-environment measures to be in Scotland?

Restore and protect riverbank vegetation damaged by historic grazing and poaching.

- ☐ Rare or non-existent
- ☐ Not very common
- ☐ Neither common nor uncommon
- ☐ Common
- ☐ Very common
- ☐ Don't know/not sure

Convert arable land at risk for erosion or flooding to low-input grassland.

- ☐ Rare or non-existent
- ☐ Not very common
- ☐ Neither common nor uncommon
- ☐ Common
- ☐ Very common
- ☐ Don't know/not sure

Establish and maintain rural sustainable drainage systems.

- ☐ Rare or non-existent
- ☐ Not very common
- ☐ Neither common nor uncommon
- ☐ Common
- ☐ Very common
- ☐ Don't know/not sure

Retain winter stubbles until early spring.

- ☐ Rare or non-existent
- ☐ Not very common
- ☐ Neither common nor uncommon
- ☐ Common
- ☐ Very common
- ☐ Don't know/not sure

Establish grass strips within or at the edges of arable fields (minimum width 3 metres).

- ☐ Rare or non-existent
- ☐ Not very common
- ☐ Neither common nor uncommon
- ☐ Common
- ☐ Very common
- ☐ Don't know/not sure

Reduce sheep stocking rates on moorlands.

- ☐ Rare or non-existent
- ☐ Not very common

- Neither common nor uncommon
- Common
- Very common
- Don't know/not sure

Remove, lower or breach embankments to restore floodplains.

- Rare or non-existent
- Not very common
- Neither common nor uncommon
- Common
- Very common
- Don't know/not sure

Establish water margins (minimum width 3, 6 or 12 metres for burns, streams and lochs).

- Rare or non-existent
- Not very common
- Neither common nor uncommon
- Common
- Very common
- Don't know/not sure

Create and manage hedgerows.

- Rare or non-existent
- Not very common
- Neither common nor uncommon
- Common
- Very common
- Don't know/not sure

Provide alternative drinking water sources for livestock.

- Rare or non-existent
- Not very common
- Neither common nor uncommon
- Common
- Very common
- Don't know/not sure

Click the 'Next' button to continue...

Do you think the implementation of these agri-environment schemes should be <u>increased</u> across farms in Scotland?

Establish and maintain rural sustainable drainage systems.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Remove, lower or breach embankments to restore floodplains.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Restore and protect riverbank vegetation damaged by historic grazing and poaching.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Convert arable land at risk for erosion or flooding to low-input grassland.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Reduce sheep stocking rates on moorlands.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Establish water margins (minimum width 3, 6 or 12 metres for burns, streams and lochs).

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Establish grass strips within or at the edges of arable fields (minimum width 3 metres).

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Create and manage hedgerows.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Provide alternative drinking water sources for livestock.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Retain winter stubbles until early spring.

- Strongly support
- Support
- Neither for nor against
- Against
- Strongly against

Click the 'Next' button to continue...

Your perception of the overall environmental benefits of agri-environment schemes

Please rank the 10 agri-environment schemes from the ones you perceive to have the greatest overall benefit at the top to the measures that you think are likely to have the least overall environmental benefits at the bottom.

In your judgement, please try to consider the overall benefits measures might offer for diffuse pollution, flooding and biodiversity that you have already considered in the previous exercise, but also other possible benefits, such as carbon sequestration, fishery enhancement and cultural benefits such as health and well-being and landscape aesthetics.

- ☐ Remove, lower or breach embankments to restore floodplains.
- ☐ Establish and maintain rural sustainable drainage systems.
- ☐ Convert arable land at risk for erosion or flooding to low-input grassland.
- ☐ Reduce sheep stocking rates on moorlands.
- ☐ Restore and protect riverbank vegetation damaged by historic grazing and poaching.
- ☐ Retain winter stubbles until early spring.
- ☐ Create and manage hedgerows.
- ☐ Establish water margins (minimum width 3, 6 or 12 metres for burns, streams and lochs).
- ☐ Provide alternative drinking water sources for livestock.
- ☐ Establish grass strips within or at the edges of arable fields (minimum width 3 metres).

[The ten measures could be dragged over to the right side of the screen to be ranked and rearranged]

Click the 'Next' button to continue...

What other measures, funded or unfunded, would you rate as very useful in providing multiple ecosystem service benefits?

[…]

Click the 'Next' button to continue...

As a final question, we would be grateful if you could add some comments around the benefits or shortfalls of any specific agri-environment measures as well as more general comments on which factors limit the uptake of agri-environment schemes in Scotland.

[…]

Click the 'Next' button to continue...

Thank you very much for completing the survey!

Your data will be analysed once all other participants have entered their survey and the research data will be kept anonymously and will not be identifiable in any publication. The research may be published in an academic journal, feel free to contact Kathleen Stosch if you would like to be informed if and when the work will be published.

We are hoping that the answers you have given will help to identify management options which are most beneficial to different stakeholder groups in Scotland and will help towards maximising the environmental benefits Scottish catchments provide in the future.

If you have any questions or comments about the research study at any point, please feel free to contact Kathleen Stosch via phone at 07766800934 or email at kathleen.stosch@stir.ac.uk