

Nature based solutions in agricultural landscapes: an ecosystem service perspective

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Abstract

Nature-based solutions (NbS) are interventions that work with, manage, or restore natural processes to address societal challenges such as climate change, biodiversity loss, and sustainable resource management. They are promoted for their potential to deliver multiple ecological, social, and economic benefits, yet their effectiveness in agricultural landscapes remains uncertain. This thesis examines the ecological performance and social dimensions of NbS, focusing on rural sustainable drainage systems (RSuDS) and beaver-influenced habitats, to assess contributions to natural capital, ecosystem service delivery, disservices, and barriers to uptake.

Beavers were shown to make a substantial contribution to Scotland's woodland natural capital. Through expert elicitation and integration into the Natural Capital Asset Index, results highlighted how beaver activity enhances regulating services at a national scale, with benefits expected to increase as populations expand. Complementary analysis of RSuDS using the Rapid Assessment of Wetland Ecosystem Services (RAWES) framework and spatial methods revealed that these interventions primarily deliver regulating services, especially water quality improvement and flow regulation, while also exposing trade-offs, limited provisioning benefits, and methodological gaps in cultural service valuation.

Further investigation into greenhouse gas disservices from agricultural ponds, using direct measurement and structural equation modelling, showed that all ponds acted as net emitters of methane and carbon dioxide, with methane exerting the greatest climate forcing. Emission drivers were linked to depth, nutrient status, and oxygen availability, offering pathways for mitigation. In parallel, a national survey of farmers and advisers shed light on the social and governance dimensions of NbS. Financial risk, inflexible design, and limited advisory support were identified as key barriers to uptake, with recommendations for simplified applications, outcome-based payments, and farmer-led collaboration.

Together, these studies provide an integrated ecological and social assessment of NbS in UK agricultural landscapes. By linking natural capital accounting, ecosystem service evaluation, and stakeholder perspectives, the thesis advances understanding of multifunctionality, disservices, and governance challenges, offering evidence to inform the design and scaling of more effective, inclusive NbS.

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List of abbreviations

AES – Agri-Environmental Schemes

AIC – Akaike Information Criterion

CAP – Common Agricultural Policy

CICES – Common International Classification of Ecosystem Services

CO₂ – Carbon Dioxide

CH₄ – Methane

CREW – Centre of Expertise for Waters

DEFRA – Department for Environment, Food and Rural Affairs

DM – Decision Maker (aggregate of expert judgments)

DO – Dissolved Oxygen

ELMS – Environmental Land Management Scheme(s)

GHG – Greenhouse Gas

IPBES – Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

MEA – Millennium Ecosystem Assessment

NCAI – Natural Capital Asset Index

NbS – Nature-based Solutions

RAWES – Rapid Assessment of Wetland Ecosystem Services

RSuDS – Rural Sustainable Drainage Systems

SEJ – Structured Expert Judgement

SEM – Structural Equation Modelling

SuDS – Sustainable Drainage Systems

TDN – Total Dissolved Nitrogen

TDP – Total Dissolved Phosphorus

TDC – Total Dissolved Carbon

Declarations and statements

I, David Bryan, wish to submit this thesis in accordance with the University of Stirling research degree regulations. Ethical approval was granted for all research reported in this thesis under GEUP reference numbers; 17793, 16176, 13209, 6766

I declare that the thesis embodies the results of my own research and was composed by me. Where appropriate, I have acknowledged the nature and extent of work carried out in collaboration with others. All research materials, sources, and contributions from collaborators have been duly acknowledged and cited. I consent to the use of plagiarism detection software for the purpose of assessing the originality of this thesis.

This thesis includes sections where artificial intelligence (AI) tools were used to assist with improving language clarity, sentence structure, and overall readability. AI tools were also used to provide technical assistance in coding. At all times, I reviewed, edited, and determined the final content of the thesis. All academic interpretation, analysis, argumentation, and conclusions are entirely my own. I confirm that this thesis represents my original work, except where due acknowledgment has been made.

Publications arising from this thesis

Chapters 3 and 4 have been submitted for publication with *PLOS One* and *Inland waters*, respectively. Chapter 2 will be submitted for publication once all the co-authors and collaborators have commented on the final draft. Chapter 5 has generated impact in a different format by taking a policy focussed approach. A policy brief (appendix H) and infographic (appendix I) have been created along with a blog post that will be distributed through Scottish government and the wider farming community. The doctoral candidate (David Bryan) was the lead author on all publications and author contributions for each publication are detailed at the beginning of each chapter.

In addition, I am a co-author on two other papers; *Towards improved accounting and mitigation of greenhouse gas emissions from ditches and canals* and *Transforming landscapes through ecosystem engineering: implications for biodiversity and ecosystem function at different scales* that are currently under review with *Environmental Research Letters* and *Landscape Ecology* respectively.

1.Introduction

1.1 Agriculture, environmental challenges, and the role of nature-based solutions

In the aftermath of World War II European governments prioritised food security. Resources were invested into modernising agriculture along with policies to incentivise increased farm outputs and affordable food for all (Billen et al., 2021). Farms rapidly became mechanised with the uptake of agricultural vehicles and engineering advances, whilst synthetic inputs were widely adopted. This marked the start of European agriculture's "green revolution", characterised by an increasing reliance on fossil fuel-powered farming and the intensive application of synthetic fertilisers, herbicides, and pesticides (Billen et al., 2021). This intensification of farming activities increased from the 1960s – 1980s with the introduction of the EU's Common Agricultural Policy (CAP) launched in 1962. Farmers were incentivised to deliver high outputs which encouraged further chemical use, larger fields, and the monocultures of certain high yielding varieties (Billen et al., 2021). From the 1980s – 2000 the environmental costs of intensive agriculture became apparent with growing evidence of pollution, habitat loss and biodiversity collapse. This generated some change with the introduction of the 1991 Nitrates Directive, and the 1992 Birds and Habitats directives along with reforms to the CAP. But intensive farming was still the norm, especially on prime agricultural land. In the 21st century we are seeing the full effects of over 70 years of increasingly intensive agricultural practices. Yields have increased fourfold (Robinson and Sutherland, 2002); however, this intensification has come at a steep environmental cost (Antonini and Argilés-Bosch, 2017).

Land-use change, largely driven by agriculture, is the most significant cause of biodiversity decline (IPBES. 2019). More than one-third of the Earth's land surface is currently used for crop cultivation or livestock grazing (IPBES. 2019). The degradation and simplification of ecosystems from agricultural pressures have been a key driver in large scale species loss and the Earth entering its sixth mass extinction event (Semenchuk et al., 2022; Raven, and Wagner., 2021). This combination of unsustainable intensification and land expansion makes modern agriculture a leading driver of global environmental change (Paz et al., 2020). Widespread declines of farmland plants, insects and birds have been observed since 1945 in the UK. This is in part due to simplification of the farming landscape with the loss of hedges, field margins and mixed habitats (Robinson and Sutherland, 2002). But it is also caused by the intensification of farming practices; Rigal et al (2023) highlighted that farmland bird populations across Europe have dropped on average 57% over 37 years with

agricultural intensification, in particular pesticides and fertiliser use, being the primary driver. The use of pesticides and herbicides is also a key driver in the widespread loss of terrestrial and aquatic insect biodiversity and biomass (Abudulai et al., 2022; Tsiafouli et al., 2015).

The intensification of agriculture is also a primary driver of land degradation and declining soil fertility. Poor management practices such as aggressive tillage and overgrazing have accelerated erosion rates beyond natural soil regeneration (Kuzyakov, Y. and Zamanian., 2019). In the EU, 24% of soils experience water erosion, especially on arable land, and soil organic carbon in croplands has fallen by ~70 million tonnes between 2009–2018 (Akca et al., 2024). The impact of intensive agriculture also impacts the hydrosphere through aquatic pollution. Freshwater and marine systems are experiencing eutrophication from nutrient runoff with a recent review highlighting that 74% of European farmland now has a nitrogen surplus that is harmful to ecosystems and human health (Akca et al., 2024). Intensive agriculture is both a victim and contributor to climate change. The current food system, which includes farming activities, land use change and the supply chain, currently accounts for around one-third of global anthropogenic greenhouse gas (GHG) emissions (Crippa et al., 2021) and is on track to rise as global food demand increases (Clark et al., 2020). At the same time, intensive agriculture is vulnerable to climate change with shifting weather patterns threatening crop and livestock productivity (Verma et al., 2025).

All these environmental challenges caused by intensive agriculture are deeply interwoven and need to be addressed simultaneously. For example, climate change exacerbates biodiversity loss (Muluneh 2021) whilst also threatening food production and security (Saleem et al., 2024). Conversely, biodiversity loss enhances climate change by impairing the ability of ecosystems to regulate carbon and climate (Weiskopf et al., 2024) whilst also undermining crop production and resilience (Dainese et al., 2019). Likewise, continued food security pressure from a growing world population contributes to further expansion and intensification of farmland (Clark et al., 2020) resulting in increased biodiversity decline and Greenhouse gas (GHG) emissions. These environmental challenges form an interlinked system that has been conceptualised in the “triple challenge” (Baldwin-Cantello et al., 2023). The three facets of this challenge are:

1. Avoid dangerous climate change by cutting GHG emissions and enhance carbon sinks, mitigation
2. Halt and revert biodiversity loss through protection and restoration of ecosystems and species
3. Secure food and nutrition for a growing population by increasing sustainable agricultural production

Recent agricultural reforms in the EU and UK have aimed to tackle this triple challenge. In the EU the CAP has been aligned with climate and biodiversity objectives (Doukas et al., 2024) whilst in the UK, England has adopted the Environmental Land Management Scheme (ELMS) and Scotland has recently introduced a new agricultural bill with reforms to the agricultural support system underway. At the core of all these policies is a nature-based solutions (NbS) approach. NbS aim to use *“Actions that protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human wellbeing and biodiversity benefits.”* IUCN (2016). NbS work with natural processes to solve problems whilst delivering co-benefits (social, ecological, economic) and promoting resiliency and adaptability. Using NbS in agriculture reframes the farmers relationship with food production and the land that they steward. Farms shift their focus from just the production of food towards a wider diversity of environmental public goods (e.g., flood protection, carbon sequestration, recreational value etc) in which food is the primary deliverable (Miralles-Wilhelm. 2023). Using a NbS approach rewrites the narrative of farming from the dichotomy of farming versus nature to farming with nature.

1.2 Using NbS to enhance natural capital and ecosystem services

A key feature of NbS is that they leverage and enhance natural capital assets. Natural capital is the combined stock of biotic and abiotic assets; they are the biophysical structures and processes that form the building blocks of ecosystems (Costanza et al., 1997). Biotic assets are the living components that include plants, animals and microorganisms whilst the abiotic assets are the non-living components such as geology, water and air. This stock of natural assets can also be renewable (e.g., freshwater, fisheries and forests) or non-renewable (e.g., fossil fuels and mineral deposits) (Costanza et al., 1997). These stocks of natural assets sustain core ecosystem functions (e.g., water filtration, nutrient cycling, primary production) which arise from the dynamic processes and interactions that occur because of the biophysical structures.

Ecosystem services are the outputs of ecosystem functions that sustain or improve human life (Cohen-Shacham et al., 2016). There are four categories of ecosystem service: 1) Provisioning services yield material products such as food or fibres; 2) Regulating services arise from the regulation of ecosystem processes such as the flood and erosion control by wetlands or pollination of crop by insects; 3) Cultural services are the non-material benefits such as recreation, aesthetic beauty, educational value and spiritual enrichment; and 4) Supporting services underpin the others by maintaining ecological conditions through soil formation, primary production, nutrient cycling and habitat provisioning. Some ecosystem service classification systems, such as the Common International Classification of Ecosystem Services (CICES), treat what were traditionally termed

supporting services as underlying ecological structures and functions that enable final services, rather than as services in their own right (Haines-Young & Potschin, 2018). Others treat them as a separate service category (Millennium Ecosystem Assessment; MEA) (De Groot et al., 2002). CICES cites double counting as the reason for this whilst MEA argues that including supporting services as a separate category prevents them from being overlooked in decision making (Haines-Young & Potschin, 2018).

Ecosystem services are translated into benefits for society when they contribute to human wellbeing. An ecosystem service may be produced but if there is no connection to human wellbeing then nobody benefits (Fisher et al., 2009). For example, a regulating service like storage of water in an agricultural pond translates into reduced flood risk so those downstream communities have greater safety and economic security, but upstream communities are not benefiting from this service. Human wellbeing is multidimensional and so these benefits can span across physical, social, cultural and psychological components (Garrett et al., 2023). Society can then value these benefits where their worth is measured or recognised. Value can be economic (market or non-market), social, cultural or ecological. For example, for a riparian woodland the non-market values may be the recreational enjoyment and aesthetic appreciation of walking through it, whilst the market value could be the lower costs of getting drinking water to safe standards because of the filtering and purifying services riparian woodlands provide.

The cascade model (Potschin & Haines-Young, 2016) (Figure 1) conceptualises the process from natural capital to societal valuation as a linear flow. It begins with biophysical structures and ecological functions, which underpin the supply of ecosystem services. These services generate benefits that contribute to human wellbeing and are ultimately assigned value within social and economic systems.

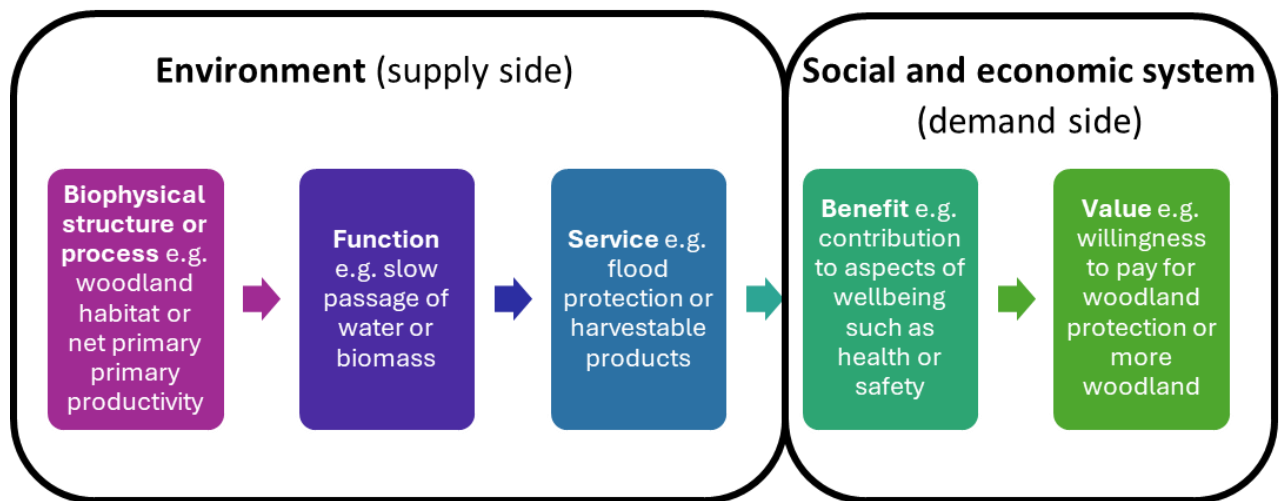


Figure 1: Cascade model adapted from Potschin and Haines-Young., 2016

Both natural capital and NbS emphasize working with nature's capacity; one by measuring and valuing it, the other by actively enhancing it. NbS works within the cascade model by expanding or improving the natural capital assets, which affects the entire chain from the ecosystem functions through to the benefits delivered.

The capacity of an ecosystem to deliver services that support human wellbeing is determined by the extent and condition of its natural capital (Natural Capital Coalition., 2016). Healthy stocks of natural capital ensure a continued and diverse flow of ecosystem services, whereas degradation undermines this capacity, leading to the erosion of both ecological integrity and human systems (Bateman and Mace., 2020). NbS offer an approach that actively builds and restores natural capital, thereby enhancing the long-term delivery of ecosystem services (Keesstra et al., 2018). In contrast, conventional grey infrastructure, such as flood walls or water treatment plants, generate a narrow range of engineered benefits without contributing to the regeneration of natural systems.

Furthermore, grey infrastructure depreciates over time, requiring maintenance or replacement, whereas NbS can improve as ecosystems mature (Zuniga et al., 2020). However, they can also degrade for instance, riparian buffer strips that initially act as pollutant sinks can become sources of nutrients if they become saturated, are poorly managed, or overloaded with runoff and sediment inputs (Stutter et al., 2019).

Through this process of ecological enhancement, NbS can create a legacy of enriched natural capital: more resilient landscapes, improved biodiversity, and replenished ecological functions that may continue to yield benefits over time. However, the long-term effectiveness of NbS is not guaranteed; many require periodic maintenance, and without appropriate management, their functionality may

decline or even reverse. Agricultural landscapes can be made more structurally diverse and ecologically robust through the integration of NbS, resulting in a higher quality and broader range of natural assets (Arnés García and Santivañez., 2021).

However, while natural capital and ecosystem services have become central to environmental research and policy, they are not without criticism. A key concern is their anthropocentric framing, which prioritises the benefits ecosystems provide to humans and may marginalise intrinsic or ethical values of nature (Schröter et al., 2014). This utilitarian perspective risks simplifying complex human–nature relationships and may overlook cultural, spiritual, or relational dimensions that are difficult to quantify. Valuation and measurement of natural capital and ecosystem services also face limitations. Simplifying complex ecological processes into discrete, quantifiable units introduces uncertainty and can omit poorly understood services, particularly cultural or supporting services (Kallis et al., 2020; Robinson et al., 2013). Monetisation, while useful for policy engagement, remains contentious because it can imply that natural and human-made capital are substitutable, which does not always reflect ecological realities (Robinson et al., 2013).

Finally, NBS although widely promoted, are subject to conceptual and practical critiques. Their broad definition can be ambiguous, potentially encompassing interventions with limited ecological benefit (Guerry et al., 2015). NbS are often presented as delivering “win–win” outcomes, yet empirical evidence highlights trade-offs and potential disservices, such as increased greenhouse gas emissions or altered disease dynamics. Effectiveness is highly context-dependent, influenced by ecological, social, and governance factors, and long-term performance requires careful design, monitoring, and adaptive management (Guerry et al., 2015; Bateman & Mace, 2020).

1.3 Co-benefits, Trade-offs and Disservices in Nature-Based Solutions

One of the primary reasons for using NbS in agriculture is their multifunctionality. NbS deliver one primary intended ecosystem service, often called an anchor service, in addition to co-benefits that can be economic, environmental and societal (Environment Agency. 2025) (Table 1). A good example of this is the creation of agricultural ponds in farming landscapes. The anchor service may enhance water management through the storage of irrigation water and buffering of drought impacts. However, their presence will also likely improve water quality by capturing pollutants in runoff (Robotham et al., 2021), regulate storm events providing downstream flood protection (Staccione et al., 2021), support biodiversity through the creation of habitat (Peskett et al., 2025) and provide an area for relaxation and recreation for the farmer and their community (Krivtsov et al., 2022). This single investment in one NbS can therefore address several challenges simultaneously, which is why

NbS are often promoted as “win-win” or “no-regrets” projects. Even if the primary goal, e.g., climate adaption, is not certain, the co-benefits ensure the effort is still valued (Raymond et al., 2017). This positive interaction where one NbS delivers multiple co-benefits is often called a synergy or bundle, where multiple ecosystem services are delivered, mimicking the multifunctionality of natural ecosystems.

Table 1: Comparison of conventional vs NbS approach and the NbS co-benefits provided in an agricultural context

ENVIRONMENTAL CHALLENGE	CONVENTIONAL MECHANISM	NBS EQUIVALENT	NBS CO-BENEFITS
Soil erosion	Terracing, contour ploughing, mechanical barriers	Cover crops, agroforestry, vegetative buffer strips	Enhances soil health, increases carbon sequestration, reduces sedimentation in waterways
Nutrient runoff (e.g., nitrogen, phosphorus)	Precision fertiliser application, nutrient management plans	Riparian buffers, constructed wetlands, catch crops	Improves water quality, supports aquatic ecosystems, reduces eutrophication
Water scarcity & inefficient irrigation	Drip irrigation, reservoirs, boreholes	Mulching, soil organic matter enhancement, wetland restoration, rainwater harvesting	Enhances drought resilience, improves water retention, recharges aquifers
Flooding & hydrological disruption	Levees, ditches, drainage tiles	River re-meandering, floodplain reconnection, wetland rewetting	Enhances flood storage, supports aquatic biodiversity, improves landscape resilience
Soil fertility decline	Synthetic fertilisers, liming, tillage	Crop rotations, composting, green manures, reduced tillage	Builds long-term soil health, reduces input dependency
Pest control	Broad-spectrum pesticides, GM pest-resistant crops	Intercropping, biological pest control, predator habitat enhancement (e.g. beetle banks, hedgerows)	Reduces chemical use, protects beneficial species, supports ecosystem stability
GHG emissions from agriculture	Feed additives, emission caps, anaerobic digesters	Silvopasture, improved grazing management, biochar, regenerative agriculture	Carbon sequestration, improved soil carbon, reduced methane and nitrous oxide
Climate change adaptation (droughts, extremes)	Crop insurance, infrastructure upgrades, climate-resilient crop breeding	Landscape heterogeneity, perennial cropping, ecosystem-based adaptation	Builds resilience, stabilizes yields, buffers microclimates

However, NbS are not always universally beneficial, sometimes you can get winners and losers. This can result in trade-offs, where the improvement in one service results in the diminishment of another (Bennett et al., 2009). For example, in an agricultural context there is often tension between provisioning services i.e. food production or crop yield, and other services (Stosch et al., 2019). Expanding or intensifying cropland can increase food production but often at the cost of regulating services like water quality, climate regulation and pest control (Power. 2010). The reverse applies too; if farmers pursue a policy of land sparing, setting aside land for biodiversity, then they reduce the agricultural area available. But if they choose land sharing, integrating biodiversity on farms, then they may have lower yields per area (Fischer et al., 2014). NbS can also generate disservices which are outcomes that have negative impacts on human wellbeing or environmental goals (Shackleton et al., 2016). In an agricultural context, farms can be affected by disservices through crop pests or disease to livestock. They can also generate disservices to the wider environment through loss of habitat and biodiversity, nutrient runoff resulting in water pollution, and pesticide poisoning of non-target species (Mancuso et al., 2021; Pe'Er et al., 2022; Wan et al., 2025).

Good management practices aim to minimise trade-offs and disservices so that the net effect of NbS is positive, generating more equitable outcomes for stakeholders and ensuring that NbS outcomes are aligned with multi-policy goals (climate, biodiversity, agriculture, water). The outcomes of NbS vary depending on the context, management, design and local environmental conditions. To achieve net positive outcomes, it is essential that synergies, trade-offs and disservices are accounted for across multiple scales (Vodden et al., 2023). Indeed, the benefits and costs of NbS shift when evaluated at different spatial scales (Raudsepp-Hearne et al., 2010). In agriculture these observed shifts are often most apparent when transitioning from the farm/field scale to the catchment/landscape scale. For example, the management choice at a field scale to install land drains might create downstream costs through enhanced nutrient loading to catchment; however, if the opposite occurred and the field was taken out of production then the farmer bears the cost whilst the benefits accrue off the farm. This can be rectified through coordinated landscape level planning or payment for ecosystem service schemes whereby spatial externalities are tied to stakeholder incentives ensuring farmers are rewarded for producing ecosystem services that others benefit from.

The temporal scale of NbS benefits also needs to be accounted for (Rau et al., 2018). The use of chemical inputs on fields (i.e. fertilizers and pesticides) may boost yields in the short term but undermine soil fertility for future generations. However, if a long-term outlook is taken through the rebuilding of soil health, then there will be long term benefits, but it may impose short term costs.

So NbS need to be evaluated over appropriate timescales to account for short, intermediate and long-term outcomes.

1.4 Rural sustainable drainage systems and beavers on farms: opportunities and challenges

Within this thesis, rural sustainable drainage systems (RSuDS) and beaver modified landscapes were chosen as exemplars for this research because they demonstrate different forms of NbS focussed on water management. Both interventions are focussed on regulating the quantity and quality of water whilst delivering co-benefits, but they differ in implementation and scale. RSuDS are planned, localized structures, whereas beaver effects are spontaneous and can extend across whole catchments. These contrasts and similarities provide a robust basis for evaluation. For example, RSuDS typically require setting aside some land for ponds or buffers but yield significant water quality and habitat gains (Kazak et al., 2022), whereas beaver activity can improve flood buffering and wildlife habitat but may unpredictably inundate farmland (Jones and Jones, 2023). Together, these complementary solutions highlight both the synergies and trade-offs of NbS: each delivers co-benefits while also imposing costs in agricultural settings (Thompson et al., 2021). In this way, examining a planned green infrastructure (RSuDS) alongside a natural intervention (beavers) provides a broad perspective on NbS multifunctionality. These complementary case studies thus provide a lens through which the multifunctionality of NbS can be explored, evaluating the multiple benefits delivered to agricultural landscapes whilst also accounting for trade-offs and disservices (MirallesWilhelm 2023).

1.4.1 Rural sustainable drainage system (RSuDS)

RSuDS is a catch-all term that covers a range of water focussed agricultural NbS (Table 2) whose anchor service is focussed on regulating the quality and quantity of water (Duffy et al., 2016). Conventional agricultural practices can contribute to diffuse pollution of receiving waters. The diffuse pollution transfer continuum (Haygarth et al., 2005) is a conceptual framework that recognises that impacts of diffuse pollution result from a combination of pollutant source, mobilisation and transfer/delivery pathways, resulting in the loading of rivers and streams with sediment, nutrients and other pollutants (Harrison et al., 2018). RSuDS intercept the transfer and delivery pathways by using natural hydrological processes to slow, spread and sink water into the landscape (Hankin et al., 2021) whilst also filtering out pollutants (Suttles et al., 2021). RSuDS also deliver a range of co-benefits from other provisioning and regulating services such as freshwater provisioning and pollination (Everard et al., 2016) through to cultural services as aesthetic beauty and recreation (Morgan and Fenner., 2019).

Table 2: (CREW, 2016; Digman et al., 2014; Ellis et al., 2012; Scottish Government., 2017)

RSuDS type	Example photo	Location within farm	Typical scale of application	Knowledge on effectiveness	Popularity / extent of uptake	Main ecosystem services delivered	Main trade-offs and limitations
Sediment bunds / traps		Field margins, downslope edge of fields	~1 m wide, ≤ 1% of field area; intercepts field edge or base of slope	Medium – well-documented for sediment capture and ~ 30–50% P reduction; less on nutrient leaching	Medium-high – widely incentivised under agri-schemes; common on slopes	Sediment retention, phosphorus capture, slight runoff slow-down	Requires periodic desilting; may reduce land for cropping; less effective for dissolved nutrients
Swales (vegetated channels)		Along contours or field drainage paths	Linear channels ~0.5–2 m wide per ha; designed along field gradient	Medium – good studies on nutrient and flow attenuation; performance tied to soil type	Medium – used in farms and roadsides; fewer formal grants	Runoff attenuation, infiltration, pollutant filtration, habitat provision	Needs space; can pond or erode if poorly maintained; less uptake on steep land
Retention ponds		Near outflows or field drainage outlets	~50–200 m ² per catchment; adjacent to field drains	High – abundant evidence of sediment and nutrient trapping; also flow control	Medium – popular in demonstration catchments; capital-intensive	Sediment & nutrient retention, flood attenuation, biodiversity support	Land take; cost/complexity; methane risk if anaerobic; maintenance burden
Constructed farm wetlands		Low-lying areas or near watercourse confluences	~200–500 m ² ; field edge or low-lying area	Medium – promising evidence for N & sediment retention; fewer GHG studies	Low-Medium – newer measure, funded in some schemes (e.g. Scotland AECS)	Sediment/nutrient capture, habitat creation, groundwater recharge	Requires management; land loss; potential for invasive species; GHG emissions risk

1.4.2 Beaver modified landscapes

Beaver modified landscapes are a more ecological but less controllable form of water management. Beavers are ecosystem engineers through the building of dams, digging of canals and felling of trees (Figure 2) (Brazier et al., 2021). They create a series of dams on streams resulting in ponds and wetland areas. The modified landscapes that beavers create function similarly to RSuDS acting as nature-driven drainage interventions. The dams raise water levels, enabling water to spread onto floodplains which slows downstream flows and increases water storage in the landscape (Puttock et al., 2021). The ecosystem services generated from this are attenuation of flood peaks and buffering of droughts (Geris et al 2023). Beaver ponds act as sinks for diffuse pollution, storing sediment and nutrients improving water quality downstream (Puttock et al., 2018). Alterations of the landscape they produce supercharges biodiversity, through the creation of a mosaic of aquatic and semi-aquatic habitats e.g. ponds, wetlands, wet woodlands, wet meadow which support diverse flora and fauna (Willby et al., 2018). Like RSuDS, beaver modified landscapes also deliver co-benefits, from provisioning services related to drinking water through the cultural services related to recreation and tourism (Thompson et al., 2021).



Figure 2: Image to the left shows an agricultural drainage ditch which has been blocked by a beaver dam which has reconnected the floodplain. Image to the right is showing a beaver dam on an agricultural pond which has raised the water levels.

1.4.3 Evaluating two key NbS as exemplars

While both RSuDS and beaver-modified landscapes aim to regulate water quantity and quality within agricultural environments, they do so via contrasting mechanisms and contexts, making them valuable exemplars for critically assessing the multifunctionality of NbS. RSuDS are deliberately designed and engineered interventions, implemented in targeted landscape positions to intercept and treat surface runoff before it enters watercourses. In contrast, beaver-modified landscapes arise

from the autonomous activities of beavers, who engineer their surroundings through dam-building and pond formation, directly altering hydrological processes within riparian zones.

These two NbS represent fundamentally different modes of implementation, one planned and modular, the other emergent and self-organising. RSuDS offer predictability and scalability, making them well-suited for integration into agri-environment schemes (AES) and broader landscape planning frameworks (O'Donnell et al., 2017; Mell, 2016). By contrast, beaver activity is site-specific and less controllable, leading to more heterogeneous outcomes that depend on the surrounding landscape context, hydrological regime, and behavioural patterns of the animals themselves (Puttock et al., 2018; Graham., 2022). These differences make RSuDS and beaver-modified landscapes particularly appropriate for comparative analysis, as they span a wide spectrum of design principles, governance challenges, and ecosystem responses within the NbS framework.

Both systems have demonstrated the potential to deliver societal and environmental benefits, particularly in relation to flood attenuation, water purification, and biodiversity enhancement (O'Connell et al., 2007; Brazier et al., 2021). However, a key challenge for both lies in their integration into existing land-use and farming systems. RSuDS may necessitate changes to field layouts or drainage infrastructure, which some farmers perceive as disruptive to productivity (Holstead et al., 2017). Nevertheless, RSuDS are typically designed to occupy minimal space, often situated within existing drainage pathways, resulting in negligible reductions in arable land (Avery, 2012). Beaver-modified landscapes, by contrast, can conflict more directly with agricultural objectives by inundating productive land, obstructing infrastructure such as drains or culverts, and causing crop damage (Campbell-Palmer et al., 2016). These tensions highlight a recurring challenge within NbS implementation on farms: navigating the trade-offs between provisioning services (e.g., food production) and regulating services (e.g., flood mitigation and water quality regulation).

A further disservice associated with both systems is the risk of 'pollutant swapping', where efforts to retain or transform certain pollutants inadvertently result in the emission of others (Freer et al., 2016). Both RSuDS and beaver ponds serve as biogeochemical sinks, capturing sediment, nutrients, and organic material. Under anaerobic conditions, however, the decomposition of this stored matter can lead to the production and release of greenhouse gases (GHGs), particularly methane (CH₄), which undermines their potential contributions to climate regulation (Lazar et al., 2014; Malerba et al., 2024). While there is emerging evidence that both systems may also sequester carbon at high rates (Gatti et al., 2018; Gilbert et al., 2021), the net climatic balance remains uncertain, underscoring the need for holistic, multi-domain evaluation of NbS performance.

What ultimately makes RSuDS and beaver-modified landscapes particularly compelling as case studies is not only their contrasting characteristics, but also their capacity to surface the tensions and synergies involved in delivering multiple ecosystem services within actively managed landscapes. RSuDS exemplify a modular, technocratic approach to NbS that fits neatly within existing environmental policy frameworks, though they remain subject to variation in long-term effectiveness and maintenance challenges (McGonigle et al., 2012). Beaver systems, on the other hand, showcase the potential for self-sustaining, process-led restoration but demand adaptive governance models due to their unpredictability and occasional stakeholder conflict (Brazier et al., 2021; Graham et al., 2020).

1.5 Human Dimensions: The Importance of Stakeholder Adoption and Social Barriers

Farmers and landowners are the primary stewards and stakeholders of agricultural landscapes. Therefore, integrating NbS into these landscapes is not just an ecological or technical challenge but fundamentally a social endeavour. Translating NbS theory into real-world implementation requires a deep understanding of how stakeholders perceive, interact with, and are supported in adopting NbS (Liu and Martens., 2023). Farmers, landowners, and advisors are the agents of change, their motivations, perceptions, and capacities ultimately determine the extent to which NbS can deliver benefits on the ground along with the policies and support (Klebl et al., 2024).

The primary mechanism through which farmers and stakeholders implement NbS is currently via public subsidies which are channelled through government led agri-environmental schemes (AES) (Hasler et al., 2022). AES financially support farmers to adopt NbS on their land that generate ecosystem services for their farm and wider society. The success of AES depends on farmer uptake at scale, which requires the farming community being sufficiently engaged and convinced (Podruzsik et al., 2024).

For many farmers the decision on whether to engage with AES is determined by practical motivations and economic viability (Bartkowski et al., 2023). In anecdotal discussions with local farmers, a common refrain is, “Can’t be green if you’re in the red,” underscoring that financial stability always takes precedence over environmental considerations (Amblard et al., 2023). Farms are, fundamentally, businesses. When AES align with farmers’ environmental values while simultaneously offering clear economic benefits, they tend to gain significant traction (Coyne et al., 2021). Using AES to financially support farmer’s uptake of NbS helps to break down the public-private benefit mismatch. Many of the benefits generated by NbS, such as improved water quality,

biodiversity enhancement, or reduced downstream flooding, constitute public goods, while the costs, including any reduction in productive capacity or increased management burden, are typically borne by the landholder (Viaggi et al., 2021). However, structural barriers within AES can undermine these efforts. AES that lack clear structure, sufficient advisory support, or that involve excessive bureaucracy, and complexity can become counterproductive (Canessa et al., 2024). When farmers are confronted with burdensome administrative requirements or poorly communicated guidance, participation may decline, and trust in the policy framework may erode (Ritzel et al., 2020).

The feasibility and long-term sustainability of NbS is also shaped by broader socio-economic contexts. Elements such as labour availability, access to capital, and farm business models influence whether NbS can be practically adopted and maintained (Iseman and Miralles-Wilhelm., 2021). Land tenure and governance structures further complicate implementation dynamics. In the UK, around one-third of agricultural land is managed by tenant farmers (DEFRA, 2022), many of whom operate under short-term or insecure tenancies. This can discourage long-term investments in NbS since tenants may not remain long enough to see the benefits (Adenuga et al., 2021). At the same time, landowners may be hesitant to authorise interventions that alter land use due to concerns about land value depreciation, the reversibility of changes, or future limitations on alternative uses (Ruto and Garrod., 2009).

A key dimension influencing the uptake of NbS in agriculture is stakeholders' understanding and trust, which is closely tied to the visibility of tangible results. However, a persistent barrier is the lag time before improvements in water quality become observable. Without early signs of benefit, it is challenging for land managers to believe their interventions are effective. For example, Meals et al. (2010) estimated that water-quality outcomes following best management practices can take from months to decades, depending on the pollutant, landscape, and hydrological pathways, with phosphorus often requiring years or longer to register measurable reductions. AES uptake among farmers is also affected by the enabling environment which includes: supportive policies, access to information, and the availability of assistance (Ferreira et al., 2020). Following on from this is the need for clear communication of the benefits of NbS. Misconceptions or a lack of accessible, trusted information presents a significant barrier to uptake (Rust et al., 2022). However, this can be addressed through targeted engagement and opportunities for shared learning (Barghusen et al., 2021). Trust is central to this process: farmers are more likely to consider new approaches when information is delivered by sources they respect, such as fellow farmers or advisors familiar with local conditions, and when their own experiential knowledge is valued (O'Sullivan et al., 2022).

Trust and knowledge exchange are strengthened through reciprocal engagement, where local knowledge is integrated into the design and implementation of AES and NbS, fostering shared ownership and more context-sensitive interventions. When farmers and landowners are involved early and meaningfully in AES design and NbS implementation, it builds trust, nurtures ownership, and increases the likelihood of long-term success (Naumann et al., 2023). For instance, co-design approaches in rural water-management schemes have been shown to produce tailored, acceptable solutions by combining scientific and local knowledge, thereby reducing conflict and improving uptake of AES (Slinger et al., 2023). Furthermore, inclusive stakeholder engagement has been shown to enhance farmers' pro-environmental behaviours: collaborative governance through farmer groups increased motivation, peer learning, and broader participation in AES across Europe (Emery and Franks, 2012; Liu and Martens., 2023). Involving farmers in problem-definition and design phases consistently leads to higher adoption and contextual fit of interventions, supporting not just uptake, but sustained engagement (Paleologo et al., 2025).

Socio-cultural factors, such as whether a practice aligns with a farmer's values, identity, and sense of good farming, play a critical role in the adoption and uptake of NbS (Dessart et al., 2019). Farmers' decision-making is shaped by a complex interplay of knowledge, beliefs, and social context (Dessart et al., 2019). These perceptions are influenced by past experiences, trusted information sources, and broader personal and community worldviews. Cultural consistency with farming traditions can significantly influence how NbS are received (Hodel et al., 2024). Farming carries a strong cultural identity, which can either support or hinder the adoption of NbS. For example, many farmers take pride in maintaining a landscape that reflects order, productivity, and care, an ideal often described in the sociological literature as the "good farmer" identity (Cusworth and Dodsworth., 2021). Within this framework, practices that allow land to appear "wild" or less intensively managed may be misinterpreted as signs of neglect or poor husbandry. This perception can generate cultural resistance if NbS visibly challenge established norms about what productive and responsible farming should look like. However, NbS that acknowledge and respect farming identities, while demonstrating how NbS can align with those values, are more likely to be accepted and sustained over time (Mikołajczak et al., 2022).

1.6 Research gaps and thesis rationale

Despite growing momentum around the promotion and uptake of NbS in agricultural landscapes, significant knowledge gaps remain at scientific, policy, and practical levels. While both RSuDS and beaver-modified landscapes are gaining attention for their delivery of multiple ecosystem services, their integration into productive farming systems remains underexplored.

1.6.1 Gaps in Scientific understanding

There is a clear need for more empirical research to delineate the conditions (e.g. environmental, climatic, landscape) under which NbS deliver their intended outcomes. Seddon *et al.* (2020) note that NbS' potential benefits have not been rigorously assessed, raising concerns about their reliability, cost-effectiveness and resilience under changing climatic conditions. Chausson *et al.* (2020) similarly found that although many studies report co-benefits between climate adaptation and broader ecological or social outcomes of NbS, very few provide integrated evaluations of trade-offs or costs. Assessments often focus narrowly on one goal without quantifying how interventions affect other ecosystem services or local communities. This suggests a pressing need for holistic, system-wide studies that monitor multiple indicators (hydrology, water quality, productivity, socioeconomic impacts, etc.) simultaneously, and across contrasting contexts.

The performance of RSuDS and beaver-modified landscapes is highly context-dependent. Factors such as soil type, hydrological regime, landscape design and species behaviour can dramatically alter performance. These nuances underscore the need to understand *when*, *how* and *where* NbS work best (i.e., spatial targeting), not just *whether* they work. For instance, Dunlop *et al.* (2024) show that NbS research has overwhelmingly targeted climate and biodiversity outcomes, while other societal benefits such as economic development, human health, food and water security, are seldom the primary focus. This uneven evidence base leaves large knowledge gaps in how NbS interventions affect livelihoods, equity and wellbeing in different settings.

Few studies have systematically quantified the net delivery of ecosystem services versus disservices over time. Critical trade-offs are often ignored; for example, NbS can create co-benefits (carbon storage, recreation, habitat) but also impose costs (reduced crop yields, higher management needs, altered water availability or greenhouse-gas emissions). Seddon *et al.* (2020) emphasize that even well-known NbS (like afforestation) can have unintended outcomes and call for rigorous accounting of both gains and losses. Likewise, Chausson *et al.* (2020) highlight that very few intervention studies compare NbS to engineered alternatives in a cost–benefit framework. For example, constructed or restored wetlands may improve water quality, but they also often emit methane and may require land that could otherwise be farmed. Such trade-offs remain poorly quantified. Overall, the lack of

longitudinal, comparative data makes it hard to judge whether NbS truly provide “net” benefits under realistic conditions (Roe et al., 2021). Long-term and landscape-scale effects are especially uncertain (Roe et al., 2021). Most NbS projects have been monitored for only a few years, if at all, and typically at small (plot or catchment) scales. It is not clear how benefits accumulate or degrade over decades. Dunlop *et al.* (2024) stress the need for long-term field trials and monitoring to identify the boundary conditions of NbS success and to capture delayed trade-offs under climate stress.

1.6.2 Gaps in Policy applications

NbS integration into policy and decision-making frameworks remains limited and fragmented. Although international strategies increasingly promote NbS to achieve multiple objectives this high-level support has not yet translated into widespread operational implementation (Dunlop et al., 2024; Hölscher et al., 2023).

One of the primary policy-related barriers is the limited and uneven evidence base on the effectiveness of NbS compared to conventional, engineered alternatives. While numerous studies report positive outcomes, few provide rigorous, context-specific evaluations of long-term performance, economic efficiency, or system resilience. The absence of comparative cost-effectiveness studies and performance metrics undermines policy confidence in NbS, particularly in high-stakes sectors such as infrastructure, water management, and agriculture (Seddon et al., 2020; Chausson et al., 2020; Linnerooth-Bayer et al., 2023).

A related constraint lies in the economic appraisal frameworks commonly used to guide investment and policy decisions. These frameworks typically focus on narrowly defined, short-term outputs, making them poorly suited to capture the multi-functional and long-term benefits that NbS provide across sectors and stakeholder groups. As a result, NbS are frequently undervalued in formal assessments, particularly when co-benefits, such as mental health improvements, social cohesion, or biodiversity gains, are difficult to monetise (Wild, Henneberry, and Gill, 2017). There is a need for more integrated valuation approaches, such as multi-criteria analysis and frameworks based on pluralistic values (Pascual et al., 2017), to ensure a more comprehensive understanding of the material and non-material benefits NbS can deliver.

Governance structures also present a significant obstacle to mainstreaming NbS. In many jurisdictions, public administration is fragmented along sectoral lines, with environmental, agricultural, water, and infrastructure agencies operating in isolation. This siloed governance is ill-suited to the cross-sectoral nature of NbS, which often require coordination across policy domains and administrative boundaries (Hölscher et al., 2023). As a result, the multi-benefit potential of NbS

is rarely fully exploited, and interventions are implemented as isolated pilot projects rather than integrated components of broader planning strategies. Furthermore, policy coherence and coordination remain insufficient. Because NbS span multiple governance levels and often require collaboration between actors with divergent interests or mandates, their implementation can be delayed or blocked when institutional responsibilities are unclear. For instance, adaptation measures such as wetland restoration may fall between agencies responsible for agriculture, biodiversity, or flood risk management, with no single body accountable for delivery (Linnerooth-Bayer et al., 2023).

Moreover, institutional inertia and what has been termed ‘grey infrastructure path dependency’ further constrain the policy space for NbS. Decision-makers, engineers, and planners often default to conventional infrastructure solutions due to entrenched norms, regulatory frameworks, and perceived risk-aversion. These defaults are reinforced by procurement procedures, standards, and financing mechanisms that favour engineered interventions, making it more difficult for NbS to compete for public or private investment (Martin et al., 2025; Linnerooth-Bayer et al., 2023).

Knowledge and capacity gaps among policymakers and practitioners also contribute to under-implementation. Many public agencies lack the technical expertise, institutional guidance, and planning tools needed to design, implement, and evaluate NbS. In the absence of robust knowledge products and shared learning platforms, even well-intentioned policies may fall short of supporting NbS at scale (Martin et al., 2025). The issue of finance presents another key challenge. NbS are widely recognised as being underfunded, despite their potential to deliver high public value. Their reliance on long-term, distributed benefits and cross-sectoral outcomes complicates financing through traditional market or public funding channels. Without dedicated funding instruments and enabling regulatory mechanisms, NbS often struggle to move beyond small-scale demonstration projects (Seddon et al., 2020).

1.6.3 Practical constraints

Practical barriers continue to hinder the effective implementation and scaling of NbS in agricultural and rural contexts. These barriers span technical capacity, financing, stakeholder engagement, monitoring, and knowledge exchange.

A major constraint lies in the limited technical expertise and ecological design capacity available among local authorities, land managers, and community-based organisations. Unlike standard grey infrastructure, NbS require context-sensitive planning informed by local ecological processes, socio-economic conditions, and landscape dynamics (Martin et al., 2025). However, this specialised

knowledge is often unevenly distributed or lacking altogether, resulting in ad hoc implementation or missed opportunities for co-benefit optimisation (Cohen-Shacham et al., 2016). Network assessments have identified awareness-raising and capacity-building as among the most pressing needs, particularly in rural areas where practitioner confidence in applying NbS remains low (European Environment Agency, 2023). Compounding this issue is the absence of consolidated technical guidance. Best practices for NbS design and maintenance are still fragmented across sectors and geographies. In many instances, projects proceed without reference to standardised protocols or long-term operational plans, undermining their resilience and replicability (Corgo et al., 2024).

Monitoring and evaluation represent another underdeveloped area. Many NbS projects do not include systematic or long-term monitoring of outcomes. The lack of standardised metrics, low-cost tools, and accessible frameworks makes it difficult to demonstrate effectiveness, build stakeholder trust, or justify continued investment (Dunlop et al., 2024). This undermines adaptive management and limits institutional learning, particularly in dynamic landscapes where service delivery may fluctuate with ecological or climatic conditions.

Financial and logistical challenges further restrict the uptake of NbS. Maintenance requirements such as vegetation control, invasive species management, or infrastructure upkeep are often underestimated during planning. Unlike conventional infrastructure, which is typically treated as a one-off capital investment, NbS demand ongoing stewardship, yet existing funding frameworks rarely account for these recurring costs (Seddon et al., 2020). Moreover, practitioners frequently lack access to blended financing models that combine public, private, and community sources. The absence of documented financial pathways contributes to uncertainty and risk aversion among implementers (McQuaid et al., 2021).

Stakeholder engagement is a critical, yet often under-supported, dimension of NbS implementation. Given that these interventions frequently aim to deliver public goods, their long-term success depends heavily on community acceptance and involvement (Baker et al., 2025). However, many NbS projects fail to incorporate structured engagement strategies or to clearly communicate the socio-economic trade-offs involved (Walker et al., 2024). Objectives such as equity and inclusion are not automatically realised through NbS and must be actively designed and facilitated (Mugari et al., 2025).

In practice, engagement is often superficial or poorly sustained. Limited planning and follow-through are common, with engagement typically confined to these early or closing stages. Many initiatives lack the time, funding, or practitioner capacity to support ongoing dialogue with local communities

(Brill et al., 2022). Practitioners often rely on ad hoc participatory exercises, with much of the long-term value lost by the cessation of projects when funding ends (Bogatinoska et al., 2022). As a result, stakeholder processes frequently become a box-ticking exercise rather than a meaningful foundation for co-creation (Mugari et al., 2025).

One approach to addressing these shortcomings is a more explicit focus on co-design, particularly with farmers and land managers. Practical efforts to engage farmers often include the formation of local advisory groups or cooperatives, which facilitate peer-to-peer learning and collaborative management. Evidence suggests that such cooperatives can significantly increase the adoption of sustainable practices by pooling technical knowledge and infrastructure, thereby enabling the implementation of environmental interventions at scale (Zhang et al., 2024). Farmer-focused co-design workshops offer effective ways to incorporate local ecological and agronomic knowledge into NbS planning. These approaches help ensure that interventions align with on-the-ground realities. For example, virtual participatory mapping has been successfully applied to co-create local NbS action plans that reflect farmer perspectives in the siting and design of interventions (Schröter et al., 2023). Moreover, involving farmers in the evaluation of trade-offs can enhance trust and long-term stewardship. Participatory modelling efforts have demonstrated that shared decision-making not only increases farmer buy-in but also fosters a sense of collective responsibility for ongoing maintenance, ultimately improving the longevity and effectiveness of NbS (Giordano and Pagano, 2023).

Finally, scaling NbS remains a persistent challenge. While many pilot projects demonstrate promising outcomes, mechanisms for knowledge transfer and replication at scale are often underdeveloped. Best practices are frequently embedded in project reports or academic publications, which may be inaccessible or impractical for practitioners to apply directly (Dunlop et al., 2024). Moreover, peer-to-peer learning platforms and open-access knowledge systems remain limited in scope, contributing to inefficiencies and missed opportunities for innovation diffusion.

Some programme-level initiatives have attempted to address this gap by embedding social learning and practitioner engagement into their design. For example, the Demonstration Test Catchments (DTC) programme in England explicitly positioned itself as a collaborative platform linking researchers, policymakers, and farm advisors with local farmers. Closely integrated with the Catchment Sensitive Farming initiative, DTC sites functioned as living laboratories where on-farm trials could be observed, discussed, and replicated by neighbouring land managers, facilitating the co-development and peer exchange of diffuse pollution mitigation strategies (Old et al., 2020). A similar approach underpins the Irish Agricultural Catchments Programme (ACP), which works with

over 300 farmers across six catchments. The ACP pairs each catchment with dedicated farm advisors who deliver both tailored support and regular group meetings. These forums enable farmers to compare experiences, build trust, and learn from each other, while on-farm demonstration plots provide tangible examples of successful interventions. Evaluation of the ACP has highlighted this combination of one-to-one advice and peer learning as a highly effective model for scaling environmental practices within working landscapes (Shortle and Jordan., 2018).

1.7 Positionality

Positionality is a person's location within their research relative to their social, cultural and disciplinary background. It is an important component of ensuring transparent and rigorous environmental research (Berger, 2015; Shaw et al., 2020). Positionality determines how research questions are framed, how data are gathered and interpreted, and the relationships that develop with participants and stakeholders (DeMarrais & Lapan, 2020). Acknowledging positionality supports critical reflection on potential biases, insider knowledge, and power dynamics, particularly in interdisciplinary and applied contexts such as ecosystem services, NbS and agricultural landscapes.

In the empirical chapters of this thesis, positionality varied according to research activity and context. For the structured expert elicitation in Chapter 2 and the survey of multifunctional farm wetlands in Chapter 3, I held a semi-insider position: I had prior engagement with conservation practitioners, beaver management networks, and landowner communities. This insider status facilitated access to expert participants and landholders, enhanced contextual understanding, and supported the co-production of knowledge grounded in local ecological and management realities. However, insider positionality also carries the risk of reinforcing prevailing narratives or skewing engagement toward familiar viewpoints (Yip. 2024). To mitigate this, structured elicitation protocols and systematic field methods were used to ensure that diverse perspectives and sites were included.

In the greenhouse gas emissions study (Chapter 4), my role was more akin to an outsider in relation to agricultural landowners and pond management contexts. This relative distance reduced assumptions about farm-level practices and encouraged adherence to standardised measurement protocols, but it also necessitated intentional efforts to build trust and negotiate access. In the agri-environmental schemes study (Chapter 5), positionality again blended insider and outsider features: I possessed background knowledge of agricultural policy and networks, yet the national survey deliberately engaged a wide range of farmers and advisers to capture experiences beyond close professional circles. This approach prioritised representativeness and broadened empirical validity, while acknowledging that policy familiarity could shape interpretation.

1.8 Aims and objectives

Addressing the uncertainties identified across a spectrum of research gaps in scientific, policy, and practical understanding requires an integrated and multidisciplinary approach that mixes ecological measurement, ecosystem service assessment, and stakeholder engagement. This thesis aims to assess the ecological and social dimensions of nature-based solutions within agricultural landscapes, with a specific focus on RSuDS and beaver-influenced landscapes, to advance understanding of how these systems contribute to natural capital uplift, deliver ecosystem services, generate potential disservices, and face barriers to adoption. The goal is to inform the design, implementation, and scaling of multifunctional nature-based solutions.

This overall aim is addressed through four specific objectives:

- Objective 1: Assess how beaver activity modifies natural capital assets and ecosystem service provisioning relative to baseline conditions.
- Objective 2: Quantify the range of ecosystem services and disservices provided by RSuDS, with particular attention to multifunctionality and trade-offs.
- Objective 3: Investigate greenhouse gas emissions from pond ecosystems, identifying key drivers of fluxes and contextualising emissions of this disservice.
- Objective 4: Characterise sociodemographic, perceptual, and structural barriers that influence the uptake of nature-based solutions among farmers and land advisors and identify opportunities to enhance adoption.

1.9 Thesis structure

This thesis comprises four data chapters, following this introduction. Chapter two explores the interaction between beavers and natural capital. Using expert elicitation, the study assessed how beaver activity influences natural capital and associated ecosystem services. These insights were then incorporated into a natural capital model to quantify the societal value of beavers, both presently and under future scenarios. This research was conducted in collaboration with NatureScot.

Chapter three focuses on the ecosystem services provided by rural sustainable drainage systems (RSuDS). A semi-quantitative field survey was used to evaluate 60 RSuDS sites, followed by spatial analysis to examine how site and landscape characteristics influence service provision.

Chapter four investigates one of the key disservices identified in the previous chapter: greenhouse gas emissions. Water, sediment, and gas samples were collected monthly from 18 ponds between

May and September. Emission rates were calculated, and structural equation modelling was used to identify the drivers of emissions.

Chapter five shifts focus to the human dimensions of NbS. A Scotland-wide survey of farmers and agricultural advisors was conducted to examine attitudes, motivations, and barriers to participation in agri-environmental schemes and the uptake of NbS. The research was carried out in collaboration with Scottish farmers and wildlife adviser's group (ScotFWAG) to inform ongoing legislative reform, specifically, the transition in Scotland's agricultural support framework, making this chapter particularly policy-relevant.

The final chapter synthesises findings from across the thesis, highlighting the common threads and offering recommendations for policy, research, and practice to support the effective implementation and scaling of NbS in agricultural landscapes.

2. Beavers increase natural capital and ecosystem services in woodland

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Narrative author contribution statement

I led the design and development of this research, including formulating the central research questions and methodological framework. I designed the structured expert elicitation using Cooke's Classical Method, coordinated engagement with domain experts, and oversaw the data collection process. I conducted the formal analysis and was responsible for integrating the elicited results into the Natural Capital Asset Index framework to generate national-scale estimates. I also developed the computational workflow and visualisations presented in this chapter. Finally, I led the interpretation of the findings and wrote the manuscript, incorporating feedback from co-authors throughout the drafting and revision process.

Abstract

Beavers (*Castor fiber*) have been shaping Scotland's landscape since their unofficial reintroduction in 2003, and their population is now in a phase of rapid expansion. Despite this ecological success, a national-scale valuation of their effect on natural capital and ecosystem services has been lacking. This study aimed to quantify the value that beavers have contributed to Scotland's natural capital over the past 20 years. We used structured expert judgement (Cooke's Classical Method) to elicit information from 22 beaver experts and to characterise uncertainties on how beavers affect the ability of woodland to generate regulating services. Results were integrated into NatureScot's Natural Capital Asset Index (NCAI) to assess changes in the value of Scottish woodlands from 2009 to 2021. Findings reveal that beavers have had a substantial net positive impact on woodland ecosystems at a national scale. In particular, they enhance habitat provision for biodiversity, soil formation and nutrient cycling, water quality regulation, and flood mitigation through mediation of liquid flows. However, uncertainty remains for services such as pest and disease control, mediation of gas and mass flows, and aspects of water chemistry regulation, where empirical evidence is limited or mechanisms remain ambiguous. As beaver populations expand through both natural dispersal and targeted translocations, these benefits are expected to increase exponentially. This is the first national-scale application of natural capital accounting to a species reintroduction, providing critical evidence for beavers as a nature-based solution to support human wellbeing.

2.1 Introduction

Natural capital refers to the stock of biotic and abiotic assets that underpin ecological systems and human wellbeing (Guerry et al., 2015). Biotic assets include all living organisms, while abiotic assets encompass features such as soils, freshwater, and atmospheric processes that provide the conditions necessary for ecosystems to function (Peksett et al., 2023). From this stock flow ecosystem services, commonly categorised as provisioning, regulating, and cultural services (Haines-

Young & Potschin, 2018). The flow of these services depends on the integrity of the underlying assets: ecosystem degradation reduces service provision, while restoration and sustainable management can enhance it (Benayas et al., 2009). A natural capital approach therefore helps policymakers account for trade-offs and synergies among land uses, prioritising investments in ecological restoration and sustainable development (Bateman et al., 2013). In Scotland, the Natural Capital Asset Index (NCAI) tracks habitat extent and condition to assess their capacity to deliver ecosystem services and guide national environmental policy (McKenna et al., 2019).

Eurasian beavers (*Castor fiber*) were once widespread in Scotland but were extirpated 400 years ago through hunting and habitat loss (Kitchener & Conroy, 1997). A licensed reintroduction at Knapdale began in 2009, while unlicensed populations also became established on the River Tay (McEwing et al., 2015). Beavers were formally recognised as native in 2016 and granted European protected status in 2019 (NatureScot, 2019). As ecosystem engineers, they alter landscapes through tree-felling, dam-building, and burrowing (Brazier et al., 2021). In riparian and freshwater systems, these activities generate ponds, wetlands, and canals, reshaping hydrology and habitat structure (Campbell-Palmer et al., 2016). Research has largely focused on water-related and biodiversity impacts, such as flood attenuation (Puttock et al., 2017; Nyssen et al., 2011) and habitat complexity (Nummi & Holopainen, 2014). Yet most studies examine individual parameters in isolation, rather than assessing the wider suite of ecosystem services within an integrated framework (Brazier et al., 2021; Thompson et al., 2021). Consequently, uncertainties remain around the overall societal value of beaver reintroduction, including trade-offs between local costs (e.g. farmland flooding) and public benefits at the catchment scale (Oliveria et al., 2023). Existing studies also remain limited in scale. Most are site-based and rarely extrapolated to landscape, regional, or national levels, leaving the cumulative impact of widespread beaver recovery poorly understood (Law et al., 2016; Puttock et al., 2018). Long-term, catchment-scale monitoring and modelling are therefore needed to build understanding at broader scales (Brazier et al., 2021). A further gap concerns their role as a nature-based solution (NbS). Beavers are increasingly promoted as an NbS for flood mitigation, drought resilience, and biodiversity enhancement (Puttock et al., 2025). However, evidence for broad cross-sectoral benefits remains limited (Thompson et al., 2021), and the NbS framework requires demonstrable contributions to human wellbeing as well as ecological restoration (Seddon et al., 2021).

Structured expert judgement (SEJ) offers a robust means to address these gaps. SEJ formally elicits and aggregates expert assessments to produce quantitative estimates where data are scarce or inconsistent (Burgman et al., 2011; Hemming et al., 2018). It has been applied to biodiversity risk assessment (Martin et al., 2012), invasive species impacts (Roy et al., 2014), and climate-related

ecological change (Kujala et al., 2013). Applied here, SEJ allows the synthesis of diverse knowledge across disciplines to estimate beaver impacts on multiple ecosystem services. By explicitly capturing uncertainty and variability, it helps translate site-level evidence into national-scale insights, offering a more comprehensive understanding of how beavers influence natural capital.

The aim of this research was to quantify the effects of beavers on natural capital and the subsequent generation of ecosystem services. The study used SEJ to produce quantitative estimates of beaver-driven change in woodlands. Specifically, the objectives were to: (1) quantify the effect of beavers on the generation of ecosystem services in woodland habitats using SEJ; and (2) incorporate SEJ-derived estimates into the Natural Capital Asset Index (NCAI) to evaluate the value beavers provide.

2.2 Methods

2.2.1 *Natural capital asset index (NCAI)*

NCAI is a composite indicator that tracks the capacity of Scotland's terrestrial ecosystems to support human wellbeing through the delivery of ecosystem services. It integrates static estimates of ecosystem service potential with dynamic, annually updated measures of habitat area and condition. This study applies the NCAI framework specifically to woodland ecosystems, introducing a single refinement: the addition of a condition indicator that accounts for the ecological effects of Eurasian beaver activity. All other elements of the index structure and calculation are consistent with the methodology described by McKenna et al. (2019).

The classification of ecosystem services used in the NCAI follows the Common International Classification of Ecosystem Services (CICES) framework (Haines-Young & Potschin, 2018). At the time this study was conducted, the NCAI system was undergoing technical updates, including efforts to address known gaps in the provisioning services dataset. As a result, the analysis presented here focuses on the two service categories with the most complete and policy-relevant data coverage: regulating and maintenance services, and cultural services (Table 3). This emphasis is supported by evidence from Xu et al. (2020) and Bryan et al. (under review), who found that analogous freshwater systems, such as ponds and small wetlands, deliver ecosystem services primarily in these two categories.

Table 3: Regulating and cultural services included in NCAI.

Category	Ecosystem service
Regulating	Mediation of waste, toxins and other nuisances (by biota)
Regulating	Mediation of waste, toxins and other nuisances (by ecosystems)
Regulating	Mediation of mass flows and erosion
Regulating	Mediation of liquid flows (hydrological cycle/flood protection
Regulating	Mediation of gas/air flows (storm protection/ventilation/transpiration)
Regulating	Pollination and seed dispersal
Regulating	Maintenance of nursery populations and habitats
Regulating	pest and disease control
Regulating	soil formation and composition
Regulating	maintenance of water's chemical condition
Regulating	global, regional and microclimate regulation
Cultural	Physical and experiential interactions
Cultural	heritage, scientific and educational interactions
Cultural	symbolic, sacred and/or religious interactions
Cultural	existence and bequest

NCAI is composed of three main components (Figure 3). The first is the ecosystem service potential base, which reflects the theoretical ability of each habitat type to deliver a range of ecosystem services per unit area under optimal conditions. This value is constant over time and provides a standardised baseline across all habitats. The second component, the wellbeing base, is derived by multiplying the service potential base by the total area of woodland habitat in each year. This represents the maximum potential contribution of woodlands to national wellbeing, assuming no degradation in condition. The third component adjusts this wellbeing base using a suite of ecological condition indicators that reflect the current state of each habitat. These indicators are normalised

and combined into a composite condition score, which is used to scale the wellbeing base each year and generate the final NCAI values. The index is presented as a time series, normalised to the year 2000 (NCAI = 100). To quantify the effect of beavers on woodland natural capital, the baseline NCAI (as currently configured without a beaver indicator) was compared with a modified version that includes the new beaver condition indicator. This comparative analysis allows an assessment of how beaver presence alters the estimated capacity of woodlands to deliver ecosystem services over time.

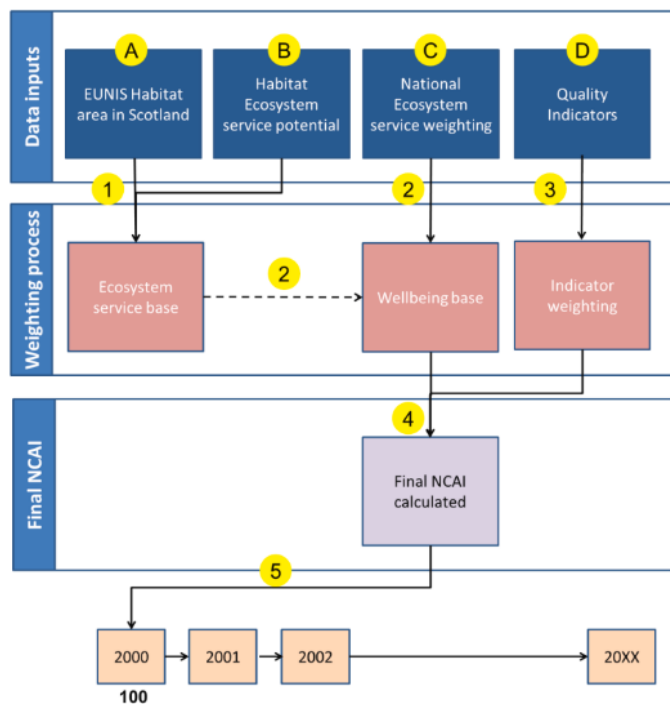


Figure 3: Flow diagram demonstrating the inputs, processes and outputs of the Natural Capital Asset Index. Letters A-D represents the four data inputs into the model. The processing of this data follows the numbers 1–5 in order. The data is weighted during the weighting process (Processes 1–3) before being compiled into the final NCAI (process 4). This process is repeated on an annual basis (process 5) to create an annual index. The number 100 refers to the baseline value of the index in the year 2000.

Woodland condition is assessed using six ecological indicators that together capture key aspects of structure, biodiversity, management, and ecosystem functioning. These include the abundance of generalist butterflies, the extent of land under agri-environment schemes, the net annual change in carbon stored in woodland biomass, the proportion of designated woodland sites in favourable condition, the total area of certified forest managed under the UK Woodland Assurance Standard, and the woodland bird index, which reflects trends in breeding bird populations associated with woodland habitats. Each indicator is updated annually using national datasets, rescaled to a common reference, and given equal weighting in the condition composite (McKenna et al., 2019).

An additional condition indicator was incorporated to reflect the growing ecological influence of beavers on woodland systems in Scotland. This indicator tracks the total area of land functionally modified by beavers, based on their surveyed territories. The most recent national beaver survey, conducted in 2020–2021, estimated the total area of beaver territories to be 10,082 hectares (Campbell-Palmer et al., 2021). The indicator was added to the existing set of condition metrics and weighted equally in the woodland composite. Given the structure of woodland types in Scotland, this indicator was only applied to those habitats where beavers are known to operate which are broadleaved deciduous and mixed (deciduous and coniferous) woodlands (Stringer et al., 2015). Conifer-dominated forests, which are typically less suitable for beaver activity, were excluded from the scope of this indicator.

The NCAI framework is designed such that each condition indicator affects all habitat within a given class. However, beaver activity is spatially constrained and does not influence all woodlands uniformly. Stringer et al. (2015) estimated that 122,876 hectares of Scotland’s woodland, specifically within deciduous and mixed types, are suitable for potential beaver colonisation. To account for this spatial variation within the constraints of the NCAI framework, broadleaf deciduous and mixed woodland classes were disaggregated into four strata: broadleaf deciduous potential beaver habitat, broadleaf deciduous remaining woodland, mixed potential beaver habitat, and mixed remaining woodland (Appendix A).

The assignment of potential beaver habitat between the broadleaf deciduous and mixed classes was based on their relative proportions of total woodland in 2021, with 89% allocated to broadleaf deciduous (109,694 ha) and 11% to mixed (13,182 ha). For mixed, which showed no change in area over the 2003–2021 period, the potential beaver habitat area was held constant across all years. In contrast, the area of broadleaf deciduous woodland increased over time, so the potential beaver area in broadleaf deciduous was backdated proportionally, following the same trend as the total broad leaf deciduous area. This allowed the beaver-relevant woodland area to increase gradually to the 2021 total, reflecting the progressive availability of suitable habitat.

The remaining woodland areas in both broadleaf deciduous and mixed were defined as those outside of the beaver-relevant subset. These followed the same temporal trends as their respective habitat classes prior to the disaggregation. Validation checks confirmed that this spatial split did not affect baseline (“no beaver”) NCAI results, ensuring methodological consistency and comparability. To reflect the spatial specificity of beaver effects, ecosystem service values derived from SEJ were applied exclusively to the potential beaver woodland strata. The remaining woodland areas were excluded from the beaver indicator, receiving no SEJ-derived values. This targeted application

ensured that the beaver condition indicator influenced only those parts of the woodland landscape where ecological effects were expected.

To represent uncertainty in the SEJ-derived values, the NCAI was run under three quantile scenarios, corresponding to the 5th, 50th (median), and 95th percentiles of expert responses. Detailed description of this is provided in section 2.3. This produced three variant time series for woodland NCAI under beaver-influenced conditions. Each was compared to the baseline no-beaver scenario to assess the direction, magnitude, and uncertainty of change in natural capital attributable to beaver activity.

To estimate annual values for the beaver indicator from the start of documented population establishment in 2009 through to the most recent survey in 2021, a deterministic exponential growth model was employed (Equation 1)

Equation 1: Model used to quantify beaver territory expansion over time from 2009 - 2021

$$A(t)=A_0 \cdot e^{rt}$$

Where t is the years since 2009, $A(t)$ = estimated beaver territory area at time t (ha), A_0 is the initial area at $t=0$, r is the exponential growth rate (year^{-1}) and e is the base of the natural logarithmic. The rate was calibrated to fit the 2021 national survey total of 10,081 hectares, assuming a 30% average annual growth rate consistent with observed expansion trends in Scotland (NatureScot., 2024) and reported values in comparable European contexts. The resulting indicator values provide a continuous annual time series of beaver-modified area from 2003 to 2021. This indicator was integrated directly into the woodland condition composite and used to update NCAI values over time, reflecting the increasing contribution of beaver-engineered landscapes to Scotland's natural capital.

This model is appropriate for populations in the early stages of colonisation, where density-dependent constraints have not yet emerged, and suitable habitat remains widely available. In such contexts, population and territory expansion are often unconstrained, resulting in exponential expansion. This pattern reflects the natural trajectory of recolonisation, in which initially low population densities and abundant unoccupied habitat allow for unconstrained expansion (Barták et al., 2013; Kawata and Ozoliņš, 2013; Petrosyan et al., 2016; Paderewski and Sienkiewicz-Paderewska, 2025).

2.2.2 Expert selection and recruitment

Experts were initially selected by the lead author through a comprehensive literature search, focusing on publications related to beavers from the preceding ten years. This approach ensured engagement with individuals actively contributing to current discourse on the topic. To expand the pool, a snowball sampling technique was subsequently employed, wherein experts who had agreed to participate were asked to recommend other suitable candidates for the study (Auster et al., 2020).

Given the geographically dispersed nature of the expert community, recruitment was conducted through online methods. Initial contact was made via email, including a concise overview of the research objectives and anticipated outputs, with an invitation for a follow-up online meeting. These meetings provided an opportunity to explore the study in greater depth, including a discussion of the research gaps being addressed, the methodological approach, and the specific role and expectations of participants. All participating experts were offered co-authorship on any resulting publications as recognition for their contributions, along with the opportunity to review final outputs. Those who agreed to participate were subsequently sent a follow-up email containing a link to begin the formal process, as well as further guidance on co-authorship and next steps. To gain an understanding of the demographic composition of our expert group they were asked two questions prior to the SEJ; firstly, what country they primarily work within; and secondly, their research fields.

2.3.3 Structured expert judgement using Cooke's classical method

We applied a SEJ approach using Cooke's Classical Model to estimate the effect of beavers on woodland ecosystem service provision. The Classical Model is a performance-based framework that quantifies uncertainty by combining expert opinions using weighted aggregation based on their statistical performance (Cooke, 1991; Cooke et al., 2007). All analyses were conducted using the Excalibur software, which implements the Classical Model algorithm (Cooke & Solomatin, 1992). The elicitation consisted of four main components: expert training, calibration (seed) questions, questions of interest, and aggregation. This approach has been widely used in ecological and environmental forecasting (Aspinall, 2010; Wittmann et al., 2015; Rongen et al., 2024). Prior to the full elicitation exercise, the entire protocol was pilot-tested with four beaver experts from the

University of Stirling. Feedback from the pilot informed minor revisions to question wording and confirmed the feasibility of the online approach. No substantial changes to methodology were required following the pilot, and the responses from pilot participants were not included in the final dataset.

Experts received online training that introduced the elicitation format, the use of uncertainty quantiles, and the conceptual basis for SEJ. The training included example questions and test exercises (Appendix B), and participants had the opportunity to request clarification. Experts were asked to provide three quantiles for each question: the 5th percentile (low bound), 50th percentile (best estimate), and 95th percentile (high bound). These define a central 90% credible interval representing their uncertainty. This format ensures all expert inputs can be interpreted as probability distributions and aggregated rigorously (Cooke et al., 2007). Training helps mitigate common issues such as overconfidence or misinterpretation of quantiles (Colson & Cooke, 2018; Hora et al., 2013).

Calibration questions, also referred to as seed variables, serve to assess experts' statistical accuracy and informativeness in estimating uncertain quantities (Cooke, 1991; Rongen et al., 2024). Experts answered 10 seed questions, drawn from the beaver literature and aligned with the ecosystem service categories in Table 3, for which the true values were known to the research team but unknown to participants at the time of elicitation. Questions were drawn from peer-reviewed literature, with care taken to ensure values were applicable across global contexts and not biased toward country-specific data. Experts provided 5th, 50th, and 95th percentile estimates for each question, followed by short written explanations of the reasoning, assumptions, and data sources underlying their uncertainty. This qualitative component ensured that experts reflected on the evidence base and made their reasoning explicit (Morgan, 2014).

Performance on calibration questions was assessed using two metrics: calibration score and information score. Calibration score quantifies how well experts' confidence intervals aligned with actual outcomes, measured using a chi-squared p-value. A higher p-value indicates better statistical accuracy, with typical significance thresholds for inclusion in aggregation ranging from 0.01 to 0.1 (Cooke & Goossens, 2008). Information score reflects the sharpness of the expert's distributions relative to a uniform distribution over a defined intrinsic range, calculated using Shannon relative information (Cooke, 1991; Colson & Cooke, 2018).

Experts can perform better on some calibration questions and worse on others, often depending on their area of expertise. In the Classical Model, this variability is accounted for in the aggregate calibration score: performance across all calibration questions is summarised into a single measure

of statistical accuracy. This score, combined with the information score, determines the expert's overall performance weight. Importantly, this weight is applied as a blanket factor across all subsequent questions of interest, rather than being adjusted for particular topics. An expert who shows strong performance on some calibration items but weak performance on others will have those differences averaged into their global weight, which governs their overall influence in the pooled estimates (Cooke & Goossens, 2008; Colson & Cooke, 2018; Hemming et al., 2018). Experts with calibration p-values below a chosen threshold were assigned zero weight in the performance-weighted aggregation, meaning their inputs were excluded from the final pooled estimates. We tested significance thresholds of 0.1, 0.05, and 0.01 to explore the effect on expert inclusion (Wittmann et al., 2015; Colson & Cooke, 2018).

Following calibration, experts were asked to assess how beavers affect the capacity of woodland to deliver a set of ecosystem services. Experts were asked questions of interest (Appendix C) which focussed on each of the ecosystem services used in CICES classification (Table 3). Each question required 5th, 50th, and 95th percentile estimates on a -1 to $+1$ scale, with values provided at fixed 0.1 intervals. On this scale, -1 represents a large negative effect, $+1$ a large positive effect, and 0 no effect. This numeric scale enabled quantification of direction and magnitude of beaver impacts on ecosystem service delivery. After each question, experts provided a brief rationale describing the information or assumptions that influenced their judgment. This transparency facilitates traceability and supports consistency across responses (Aspinall & Cooke, 2013). The entire elicitation process was conducted remotely. Experts accessed training materials, calibration questions, and target questions through an online portal. The full survey including the calibration questions and questions of interest can be found in (Appendix C)

Expert judgments on the target questions were aggregated using both performance-weighted and equal-weighted approaches for comparison. In the performance-weighted method, only experts with calibration p-values above the chosen significance level were included, and their distributions were combined using normalized weights derived from their performance scores. The resulting aggregate, referred to as the Decision Maker (DM), represents a composite distribution that reflects both the accuracy and informativeness of contributing experts (Cooke et al., 2007; Rongen et al., 2024). Performance weighting can be further "optimised" within the Classical Model, meaning that the contribution of each expert is adjusted to maximise the DM's combined calibration and information scores under cross-validation (Cooke & Goossens, 2008; Colson & Cooke, 2017). In this approach, experts with higher statistical accuracy and narrower credible intervals receive greater weight, while poorly calibrated experts may receive little or no weight. For comparison, we also generated an equal-weighted aggregate, in which all expert distributions were given the same

weight regardless of calibration performance (Wittmann et al., 2015).

Cultural services were excluded from the final decision-making analysis. This was motivated by two primary considerations. First, when cultural services were included alongside regulating services, the resulting DMs, whether equal-weighted or performance-weighted performed substantially worse for both calibration and combined scores (Appendix D). Specifically, calibration scores for these DMs were reduced by approximately an order of magnitude compared to when only regulating services were considered. Second, expert representation for cultural services was notably limited (Figure 4). Most experts came from backgrounds in biophysical and ecological domains, so we lacked sufficient coverage from social sciences or humanities, disciplines more closely aligned with cultural ecosystem services. As such, their inclusion in the aggregation risked introducing unrepresentative or poorly informed judgments.

2.3 Results

2.3.1 Respondents

Fifty-six experts were contacted with twenty-seven of those completing the SEJ (Figure 4). Most of those experts were from Scotland and England but we also had multiple experts from France, Italy and Poland (Fig 4 A). Most experts self-selected as ecologists or biologists (Fig 4 B) with several geomorphologists, hydrologists and environmental scientists. We received a small number of respondents who identified as phycologists, sociologists, economists or planners. Of the twenty-seven experts who completed the SEJ, twenty-two were taken forward to the final DM; the remaining five were excluded because they either failed to complete the elicitation in full or submitted responses incorrectly.

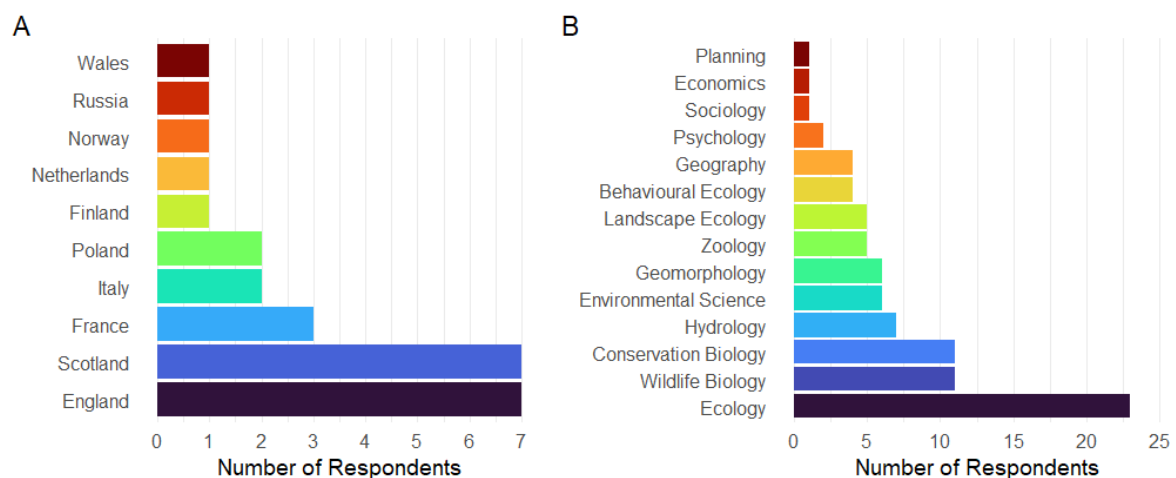


Figure 4: A shows the location of the respondents who completed the SEJ. B shows the field of research the experts self selected. Experts could select multiple fields.

2.3.2 Structured expert judgement

The equal weighted (EQ) DM achieved a calibration score and combined score higher than any of the performance weighted (GL) DMs using fixed significance levels (GL 0.01, 0.05, and 0.1) (Table 4). The only method that outperformed EQ was the optimised performance weighting (GL Opt). However, this optimisation resulted in a highly concentrated weight distribution, placing all weight on a single expert. While this produced strong statistical performance, such a strategy may compromise the robustness and legitimacy of the decision process, as it excludes the diversity of perspectives that typically justifies expert elicitation.

In terms of information scores, EQ was consistently lower than all performance-weighted methods. This is an expected outcome; performance weighting typically rewards tighter uncertainty bounds, thereby increasing information content. Nevertheless, within the SEJ framework, statistical accuracy (i.e., calibration score) is prioritized over informativeness. A highly informative but poorly calibrated distribution is more likely to exclude true values and lead to overconfident or erroneous decisions. Overall, there was considerable variability in both calibration and information scores across the expert panel (Table 4). A subset of experts demonstrated strong performance in both dimensions. Notably, Experts R, S, N, K, and V achieved the highest combined scores, ranging from 0.5861 to 0.6998. These experts exhibited well-calibrated assessments (calibration scores ≥ 0.423) alongside moderate to high information scores. Their performance indicates a robust capacity to provide both accurate and informative judgments, making them strong candidates for contributing significantly to the aggregated distribution.

Conversely, several experts displayed poor statistical accuracy, as reflected by calibration scores far below the commonly accepted threshold of 0.05. For instance, Experts J ($2.31e-05$), D (0.00043), O (0.002095), and A (0.002095) exhibited low calibration despite relatively high information scores. This pattern suggests overconfidence, in which experts provide narrow intervals that fail to capture the true values. Importantly, this does not imply that these experts lack valuable knowledge; rather, it highlights the statistical outcome of the calibration procedure. Within Cooke's Classical Model, such experts contribute less to the performance-weighted aggregation, as the method prioritises well-calibrated judgements to maintain overall accuracy of the DM (Cooke & Goossens, 2008; Colson & Cooke, 2017).

A third group of experts, including Experts C, I, and P, demonstrated moderate performance. Their calibration scores fell in an intermediate range (0.086–0.1822), accompanied by relatively high information scores. While this combination suggests a tendency toward narrower, more confident

intervals that were not always statistically well-calibrated, the Classical Model's weighting scheme accounts for this by proportionally reducing their influence. Thus, although these experts do not contribute as strongly as the top performers, they are not excluded entirely and can still add value by broadening the range of perspectives represented in the aggregate, provided their input is moderated through performance-based weighting (Cooke, 1991; Colson & Cooke, 2017).

Table 4: First section is showing the performance of the individual experts for just the regulating services. The second section is showing the performance of the different DMs just for regulating services. The third section is showing the performance of the different DMs for regulating and supporting services. EQ = equal weighting. GL = Global weighting. The number after GL is the significance level used. Opt = Optimised.

	Calibration Score	Information score		
		All	Seed	Combined score
Expert R	0.6652	0.7783	1.068	0.6998
Expert S	0.6789	0.9722	1.031	0.6996
Expert N	0.423	1.01	1.523	0.6442
Expert V	0.5332	0.6704	1.196	0.6375
Expert F	0.5332	0.9827	1.133	0.6043
Expert K	0.6652	0.7851	0.8946	0.5861
Expert I	0.1822	1.392	1.265	0.2305
Expert C	0.1822	1.205	1.227	0.2235
Expert P	0.1424	1.026	1.277	0.1819
Expert E	0.1424	0.7084	1.007	0.14434
Expert G	0.08624	0.9001	1.161	0.1001
Expert Q	0.08624	1.189	1.016	0.08758
Expert T	0.02606	0.994	1.432	0.03733
Expert L	0.02005	1.455	1.448	0.02903
Expert M	0.02606	0.8613	0.9498	0.02475
Expert H	0.02606	0.7465	0.7584	0.01977
Expert U	0.01182	1.158	1.098	0.01928
Expert B	0.0124	1.535	1.381	0.01713
Expert O	0.002095	1.406	1.831	0.003834
Expert A	0.002095	1.026	1.236	0.002589
Expert D	0.0004303	1.233	1.335	0.0005747
Expert J	2.31e-05	1.249	1.226	2.83e-05
Regulating services				
EQ	0.6552	0.2492	0.3942	0.2583
GL 0.1	0.423	0.3072	0.5646	0.2389
GL 0.05	0.423	0.3041	0.5618	0.2377
GL 0.01	0.423	0.2992	0.543	0.2297
GL Opt	0.6789	0.9722	1.031	0.6996

The performance of the equal-weighted DM against the calibration (seed) questions is illustrated in Figure 5. The results show a variable pattern of alignment between the DM's central estimates and the actual values across the seven seed questions. For five of the seed questions; Q1 (local climate regulation), Q4 (landscape biodiversity), Q5 (water quantity and mass flows), Q6 (soil formation and carbon cycling), and Q7 (aquatic habitat regulation), the DM's median estimates were relatively close to the true values, indicating a high degree of accuracy. In particular, Q1, Q6, and Q7 were

characterized by narrow uncertainty intervals with the true values situated near the medians, suggesting confident and well-calibrated judgments. Q5 (water quantity and mass flows) also performed reasonably well, though with a wider interval, reflecting a more cautious estimate that still captured the true value effectively.

Q2 (feeding habits) and Q3 (water quality and pollutants), in contrast, exhibited greater deviation between the median estimates and the true values, despite the true values still falling within the uncertainty bounds. These discrepancies indicate lower accuracy rather than a failure of calibration. Moreover, both questions featured some of the largest uncertainty intervals in the set, suggesting greater uncertainty or disagreement among experts regarding these topics. The combination of imprecise central estimates and broad ranges implies that these questions posed more difficulty for the expert panel, potentially due to knowledge gaps or inherently higher variability in the underlying phenomena.

Across all questions, the width of the uncertainty intervals varied notably. The tightest intervals were observed for Q1 (local climate regulation), Q6 (soil formation and carbon cycling), and Q7 (aquatic habitat regulation), reflecting higher expert confidence, while broader intervals in Q2 (feeding habits), Q3 (water quality and pollutants), and Q4 (landscape biodiversity) point to more cautious assessments. This variation in informativeness is consistent with differences in expert familiarity and consensus across the seed domains.

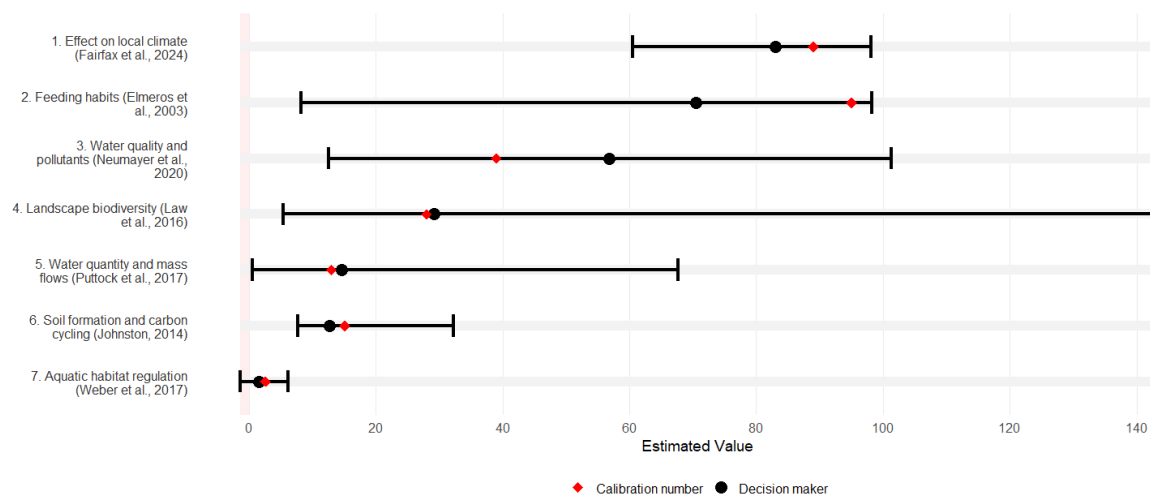


Figure 5: Caterpillar plots where the red dot is the actual number and the black dot the 50% quantile from the equal weighted DM. The upper limit is the 95% quantile and lower limit is the 5% quantile.

The results from the equal-weighted DM for the questions of interest, i.e. the effect of beaver activity on the capacity of woodland ecosystems to deliver regulating services, are presented in Figure 6. Each horizontal line represents the 90% credible interval (from the 5th to the 95th

percentile), with the central point marking the 50th percentile (median estimate) for each service. Notably, all regulating services (Table 3) assessed had their 50% quantile estimates located in the positive section, indicating a consistent expert consensus that beaver activity has a beneficial impact on the provision of these ecosystem services. The mean of the 50% quantile estimates was 0.27, reflecting a positive overall effect. Uncertainty was also characterized across services. The mean of the 5% quantiles was -0.23, while the mean of the 95% quantiles was 0.83, suggesting that while a minority of estimates allow for small potential negative effects, the central tendency remains strongly positive.

The mean range between the 5% and 95% quantiles was approximately 1.1, indicating a moderate degree of uncertainty overall. Most services exhibited similar interval widths, with two exceptions. *Mediation of mass flows and erosion* had the widest interval (1.5), indicating greater variability in expert judgments. In contrast, *mediation of waste, toxins, and other nuisances (by ecosystems)* had the narrowest interval (0.87), suggesting stronger agreement among experts for this particular service. One service; mediation of waste, toxins, and other nuisances by biota, was removed from the analysis. In the CICES framework, mediation by biota refers to regulation carried out directly by particular organisms, which in this case would mean beavers themselves. By contrast, mediation by ecosystems refers to regulatory functions of the wider habitats and ecological processes that beaver activity generates or modifies, such as wetlands filtering pollutants or soils buffering nutrient flows. A review of the qualitative justifications provided by experts revealed significant confusion between these two categories, with many responses treating them as interchangeable. To avoid redundancy and misinterpretation, mediation by biota was therefore excluded from the final analysis.

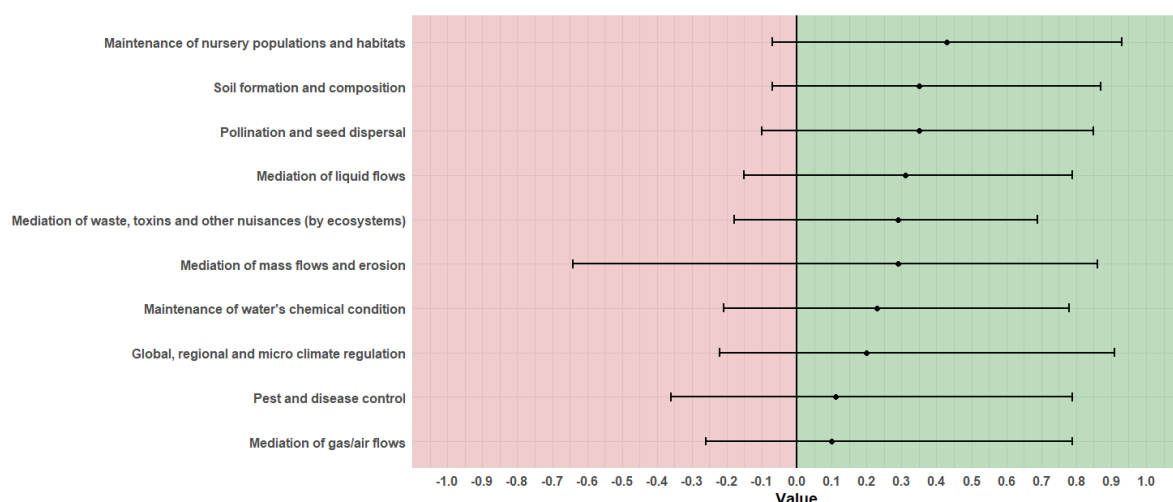


Figure 6: Caterpillar plots showing the equal-weighted DM outputs for regulating ecosystem services. Each horizontal line represents the aggregated expert distribution for a single service, with the central dot indicating the median (50% quantile)

and the whiskers spanning the 5%–95% quantile range. Wider intervals reflect greater uncertainty among experts, while narrower intervals indicate higher agreement.

In summary, experts expressed a clear and consistent belief that beaver activity enhances the regulating functions of woodland ecosystems. The concentration of central estimates in the positive domain, combined with moderate uncertainty and strong alignment across services, lends weight to the inference that beavers generally promote ecosystem service delivery in these landscapes. The full table of results from the EQ DM can be found in appendix D.

2.3.3 NCAI

Between 2009 and 2021, all scenarios of beaver effects on natural capital began at a baseline value of 100 (Figure 7). The ‘no beaver’ scenario, representing the NCAI model run without the beaver indicator, exhibited a modest upward trend over time, reaching approximately 133 by 2021. This increase is attributable to the overall expansion of deciduous woodland area across Scotland, which positively influences the NCAI metric even in the absence of beaver activity.

In contrast, the 5%, 50%, and 95% beaver scenarios represent model runs that included the beaver indicator at varying quantile estimates derived from structured expert judgement, corresponding to low, medium, and high estimates of beaver impact, respectively. The 5% beaver scenario initially mirrored the upward trend seen in the ‘no beaver’ scenario, also benefiting from increased deciduous woodland. However, as beaver territory began expanding exponentially, the weakly positive effect associated with the woodland expansion was overwhelmed. The scenario transitioned into a steady decline, dropping below the baseline value by 2010 and falling to approximately 24 by 2021.

The 50% beaver scenario, representing the central estimate, displayed steady and accelerating gains. Values rose from 100 in 2009 to about 146 by 2016 and continued increasing to over 225 by 2021. The 95% beaver scenario, representing the upper estimate of beaver impact, showed the strongest positive trend. From the baseline of 100 in 2009, values increased sharply throughout the time series, reaching over 381 by 2021, more than double the central estimate and nearly three times the value of the ‘no beaver’ scenario.

Divergence among the scenarios became increasingly pronounced after 2012. While the ‘no beaver’ scenario remained relatively stable, the 5% beaver scenario declined steadily, and both the 50% and 95% beaver scenarios showed accelerating increases, particularly in the final third of the time series. The full output from all the NCAI scenarios can be found in Appendix E.

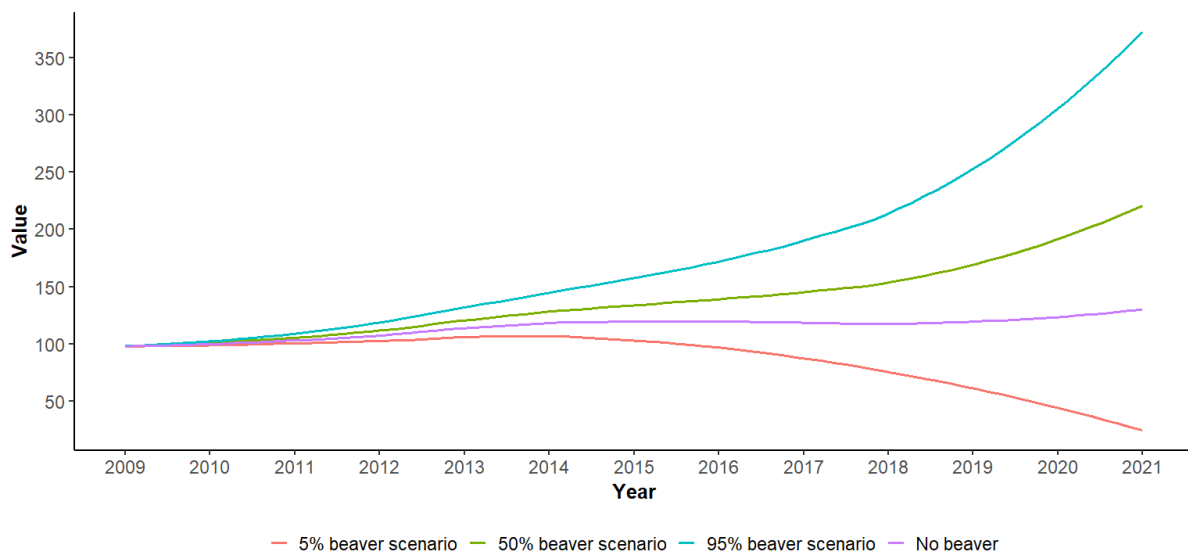


Figure 7: Line graph showing how the 5% quantile, 50% quantile, and 95% quantile estimates from the SEJ transfer into change within the NCAI. The 'no beaver' scenario, which is NCAI without the beaver indicator included, is shown for comparison.

2.4 Discussion

2.4.1 Effect of Beavers on Ecosystem Services and Natural Capital

This study provides evidence that Eurasian beavers exert a net positive influence on woodland ecosystem services. Expert elicitation revealed a strong and widespread consensus that beaver activity enhances the delivery of these regulating services, a finding made more robust by the diversity of disciplines and geographic range of the expert panel. However, despite the general agreement on direction, the confidence intervals from the SEJ were relatively broad. This variation likely reflects both the inherent complexity of ecological processes and the limitations posed by gaps in the empirical evidence base (Stringer & Gaywood, 2016). The use of equal weighting in aggregating expert estimates, adopted due to poor cross-domain calibration in performance-weighted DMs, may have further contributed to this uncertainty by preserving a wider range of expert perspectives (Colson & Cooke, 2017; Hemming et al., 2018). Nevertheless, the overall signal remains clear: beaver activity is strongly associated with enhanced ecosystem service provision, even if the magnitude of specific effects remains subject to contextual variability and further empirical validation.

Mediation of waste, toxins, and other nuisances (by ecosystems)

This service emerged as one of the most robust in the study, with consistent expert agreement and a strong supporting evidence base. Beaver-modified systems are widely recognised for enhancing the natural filtration and processing of contaminants, and the SEJ distribution showed a strong central tendency toward positive impact.

Beaver dams and wetlands slow water flow and increase residence time, promoting sedimentation and removal of pollutants such as phosphates, nitrates, and suspended solids (Bason et al., 2017; Puttock et al., 2018). Vegetation assimilates nutrients into biomass, while anaerobic sediments enable microbial denitrification (Yousaf et al., 2021). These processes are particularly valuable in agricultural catchments, where dams intercept nutrient-rich runoff and mitigate downstream eutrophication (Puttock et al., 2017).

Experts also noted limitations. Filtration efficacy may decline in large rivers or highly polluted systems, and dam breaches can release stored sediments. Potential disservices include methylmercury formation in anoxic sediments and algal blooms in nutrient-rich ponds (Błądzki et al., 2011; Larsen et al., 2021; Čiuldienė et al., 2020).

Synergistic benefits from adjacent riparian woodlands were also recognised. Beaver-driven coppicing and increased ground vegetation can stabilise banks and enhance nutrient interception (Wilson et al., 2024). Overall, despite context-specific risks, the balance of evidence and expert opinion indicates a consistently positive role for beavers in mediating waste and maintaining water quality.

Pollination and seed dispersal

This service received a high median estimate, ranking above more empirically established services such as mediation of waste. The elevated scores likely reflect the recognised role of beavers in enhancing riparian biodiversity (Stringer & Gaywood, 2016; Orazi et al., 2022). However, empirical evidence specific to pollination and seed dispersal in beaver-created habitats remains scarce (Law & Willby, 2014; Levine & Meyer, 2019), which is reflected in the wide confidence intervals recorded by experts.

Experts consistently linked beaver activity to indirect improvements in pollination. Dam building, canal excavation, and tree felling create diverse wetland mosaics that increase light penetration, support flowering plant diversity, and improve habitat connectivity (Willby et al., 2018). These changes likely facilitate pollinator abundance and diversity, including insects such as bees and butterflies, and vertebrate seed vectors like birds and small mammals (Andersen et al., 2024; Cook et al., under review). Seed dispersal was seen as a further, though more variable, benefit. Beavers

may move propagules directly by transporting woody material or indirectly by creating aquatic dispersal pathways. Willow and aspen are particularly relevant, regenerating readily from beaver-cut stems (Jones et al., 2009; Runyon et al., 2014). Experts also noted that conditions in beaver wetlands may support frugivorous birds and mammals that act as seed dispersers (Rosell et al., 2005).

Potential downsides were also raised. Dams may trap seeds that would otherwise disperse downstream, and beaver activity could aid the spread of invasive plants such as Himalayan balsam and Japanese knotweed. While herbivory may suppress some non-natives, outcomes are highly context dependent (Law et al., 2019). Several experts questioned combining pollination and seed dispersal into a single category. While both concern reproductive materials, their mechanisms and vectors differ. A disaggregated approach would likely yield more precise estimates in future assessments.

The maintenance of nursery populations and habitats

Among all regulating services assessed, this category received the highest central estimate, reflecting the well-documented role of beavers as keystone engineers that create critical habitat for a wide array of taxa (Brazier et al., 2021). Beaver activities such as dam building, tree felling, and pond creation generate structurally complex wetlands that provide ideal conditions for species during vulnerable life stages (Romansic et al., 2021; Washko et al., 2022). Experts consistently emphasised sheltered waters, woody debris, emergent vegetation, and increased edge habitat as key features supporting the growth and survival of young organisms (Nummi & Holopainen, 2014; Fedyń et al., 2022).

Multi-trophic benefits were also highlighted. Accumulated deadwood offers nesting or foraging sites for insects, cavity-nesting birds, and small mammals (Thompson et al., 2016; Sundell et al., 2021). Beaver lodges themselves may host species such as otters, bats, and rodents (Mott et al., 2013; Rosenthal, 2021), while canopy openings and coppicing enhance understory growth that benefits pollinators and ground-nesting fauna (Rossell et al., 2005).

Caution was expressed around species-specific trade-offs. Dams may hinder salmonid migration, and sedimentation can alter substrates critical for spawning fish (Fritz & Gangloff, 2022). Yet these risks were generally viewed as localised and outweighed by the wider ecological gains.

Spatial scale was seen as an important qualifier. While nursery benefits concentrate in dammed or ponded areas, hydrological change and edge effects extend their influence into adjacent habitats (Brazier et al., 2020). As noted by one expert, “beaver-modified systems are biodiversity hotspots, not just for aquatic species, but across entire landscapes.” This aligns with field evidence recognising

beaver wetlands as refugia for ecological and climate resilience (Dittbrenner et al., 2018; Fairfax & Westbrook, 2025).

Pest and disease control

This service received low median scores and wide uncertainty intervals, reflecting both conceptual ambiguity around how beaver activity influences pest and pathogen dynamics and the lack of direct empirical evidence.

Some studies suggest positive impacts. For example, reductions in *E. coli* downstream of beaver dams point to sedimentation and microbial processes limiting pathogen transport (White et al., 2025). Such effects may be mediated by physical trapping, anaerobic denitrification, and biofilm activity (White et al., 2025; Murray et al., 2023). Increased biodiversity and habitat heterogeneity may also promote ecological stability, supporting the “dilution effect” hypothesis whereby higher species richness reduces disease prevalence (Keesing & Ostfeld, 2021; Espira et al., 2022).

However, potential disservices were also identified. Beaver ponds may host parasite vectors such as gastropods that transmit trematodes, and could support pathogens including *Giardia lamblia* (Common et al., 2024; Turillazzi et al., 2024). Several experts emphasised that giardiasis is more commonly associated with livestock or human contamination, suggesting the beaver’s role is likely overstated.

Impacts on invasive species were another area of uncertainty. Beavers may disperse invasive plant fragments downstream, exacerbating spread of species like Japanese knotweed or Himalayan balsam (Stringer et al., 2016; Piętko & Misiukiewicz, 2022). Conversely, herbivory and hydrological disturbance may suppress certain non-natives, though outcomes remain highly context-specific.

Global, regional, and micro-climate regulation

This category received a moderately positive median estimate but wide uncertainty bounds, reflecting both conceptual complexity and varied impacts across spatial and temporal scales. As defined by CICES, it includes carbon sequestration, greenhouse gas (GHG) emissions, temperature buffering, and local atmospheric regulation, processes that operate differently and are difficult to evaluate as a single measure.

Experts most consistently identified local effects. Beaver wetlands were described as cooler, more humid environments that buffer air temperatures and moderate extremes through surface water, evapotranspiration, and shading (Larsen et al., 2021). Ponds also provide thermal inertia and microclimatic refuges during droughts or heatwaves (Weber et al., 2017). Increased vegetation

heterogeneity and water availability were seen as enhancing ecological resilience, with implications for species distributions and fire resistance (Fairfax & Whittle, 2020).

At larger scales, opinion was divided. Beaver wetlands can act as carbon sinks, storing organic matter in sediments (Puttock et al., 2018; Jordan & Fairfax, 2022). Yet methane emissions from new or shallow ponds were highlighted as a counteracting source of GHGs (Whitfield et al., 2015; Wohl & Inamdar, 2025). Several experts emphasised temporal dynamics: while young ponds may emit methane, long-term sediment accumulation and vegetation regrowth could offset these emissions. Tree felling was also debated, seen by some as a carbon loss, but by others as a driver of regrowth and diversification that sustains woodland carbon stocks.

Most experts stressed that beaver effects remain small at regional or global scales compared with anthropogenic drivers like fossil fuel combustion and deforestation. Some suggested broader restoration across continents could yield more significant climate effects, but at current densities, impacts are modest and primarily local.

Maintenance of water's chemical condition

Experts generally viewed beavers as supporting the regulation and stability of water chemistry in modified catchments, though confidence levels varied. This service is distinct from pollutant filtration, focusing on maintaining chemical balance, such as nutrient ratios, dissolved oxygen, and pH, rather than removing external contaminants.

Beaver wetlands often act as biogeochemical buffers, moderating nutrient fluxes through microbial activity, decomposition, and plant uptake. Enhanced biodiversity and biomass production may promote nutrient assimilation, while organic matter trapped in impoundments contributes to decomposition and cycling (Cooper et al., 2025). In this way, beaver systems can sustain chemical stability even under external pressures.

Potential imbalances were also noted. Oxygen depletion in stagnant ponds may promote eutrophication or release methylmercury from sediments (Levanoni et al., 2015). These risks were described as highly variable, influenced by pond age, succession stage, and seasonal dynamics. At the catchment scale, experts questioned whether localised improvements could offset broader pressures such as legacy nutrient loading or intensive land use. Beavers may therefore contribute positively to water chemistry, but usually as one component within larger landscape strategies (Smith et al., 2020).

Mediation of mass flows and erosion

This service showed the largest uncertainty intervals, reflecting both the complexity of sediment transport and divergent expert views on the scale at which beaver activity matters. The median estimate was positive, but expert scores spanned from stabilising to neutral or negative.

Most experts emphasised the role of dams in attenuating fluvial energy, slowing velocity, and trapping sediments, particularly in headwater streams and agricultural catchments (Puttock et al., 2018; Grudzinski et al., 2022). Beaver networks also reshape channel geomorphology: stair-step dam sequences buffer incision, increase lateral connectivity, and dissipate flow energy (Pollock et al., 2014; Westbrook & Cooper, 2024).

Localised erosion risks were widely acknowledged. These include bank collapse from burrowing, dam-edge destabilisation, and sudden sediment release during breaches (Larsen et al., 2021; Nash et al., 2021). In engineered channels or flood defence structures, such impacts can be especially problematic. Vegetation dynamics added further complexity: tree felling initially reduces bank stability, but regrowth of dense shrub and herb layers may enhance rooting and interception over time (Fesenmyer et al., 2018).

Context was repeatedly cited as decisive. In small, incised streams, beavers can substantially stabilise sediment regimes, while in large or high-energy rivers their influence is often negligible. Several experts stressed long-term resilience: although dams may cause short-term disturbance, cumulative effects generally increase channel stability and erosion buffering, consistent with the concept of “zoogeomorphological resilience” (Butler et al., 2018; Adamchak et al., 2025).

Mediation of liquid flows

This service was assessed with high confidence and consistency across experts. Beaver activity, particularly dam building, wetland creation, and canal excavation, slows surface flows, increases residence time, and facilitates groundwater recharge (Brazier et al., 2020; Geris et al., 2022). These hydrological changes reduce peak discharges during storms and sustain baseflow during dry periods, buffering extremes and enhancing catchment resilience (Puttock et al., 2021; Graham et al., 2022).

Although individual dams may fail during large flood events (Neumayer et al., 2020), experts stressed that cumulative effects of multiple structures across a network provide meaningful flood attenuation (Puttock et al., 2021). Several also highlighted aquifer recharge, where retained surface water promotes vertical infiltration (Feiner et al., 2015).

Vegetation changes were identified as complementary mechanisms. Tree felling and coppicing can increase ground vegetation and surface roughness, slowing runoff and supporting flood regulation (Larsen et al., 2021). While short-term tree loss may reduce interception, most experts argued that this is outweighed by the storage and retention capacity of beaver wetlands.

Effectiveness was considered context-dependent, varying with slope, catchment size, and substrate permeability. Dams were seen as less reliable under extreme rainfall or rapid snowmelt, but under typical conditions their influence on liquid flow regulation was consistently judged positive.

Soil formation and composition

This service received the second highest median score in the SEJ, indicating broad expert agreement that beavers positively influence soil development. Confidence intervals were moderately narrow, suggesting a shared belief in their beneficial role, though uncertainty remained around spatial and temporal variability.

Sediment retention was the most frequently cited mechanism. Dams slow water velocity, trapping fine sediments rich in organic matter and nutrients (Puttock et al., 2018; Rees et al., 2024). These sediments stabilise with vegetation, forming carbon-rich beaver meadows that persist long after abandonment (Laurel & Wohl, 2019). Floodwaters also redistribute nutrient-rich sediments across adjacent riparian zones, enhancing fertility and supporting soil development beyond the immediate stream corridor (Grudzinski et al., 2022).

Experts also noted that beaver felling and coppicing contribute woody debris and litter, stimulating microbial and fungal activity and promoting nutrient cycling (Vernon et al., 2025). Riparian zones with fluctuating water tables may be particularly important hotspots for carbon sequestration and nitrogen fixation.

Caveats included waterlogging, which can impair decomposition, alter soil communities, and reduce productivity in some contexts (Zhao et al., 2022). Experts diverged on temporal dynamics: some stressed slow, cumulative enrichment over decades, while others highlighted measurable short-term gains from organic sediment accumulation. Spatial limits were also acknowledged, effects diminish with distance from waterways and are less pronounced in upland or non-riparian soils.

Mediation of Gas flows

This service was assigned a modest positive effect but with high uncertainty, reflecting limited empirical research and a lack of specialist expertise among respondents. Experts generally suggested that beaver-engineered wetlands may influence local air flows and microclimates by expanding open

water and wetland cover. Dams and ponds enhance evapotranspiration and humidity, creating buffering effects on temperature extremes and contributing to local cooling (Larsen et al., 2021; Wohl & Inamdar, 2025). However, most stressed that these effects are spatially constrained. Beavers may shape microclimates at the site level, but their influence on atmospheric dynamics beyond immediate habitats is minimal. Several experts expressed scepticism about attributing broader gas flow regulation to beaver activity given the limited scale and evidence base.

2.4.2 Effect of beavers on natural capital

The integration of SEJ derived ecosystem service estimates into the NCAI offers insight into the potential national-scale effects of Eurasian beaver. The NCAI index trajectories clearly diverged across the beaver and no-beaver scenarios. While the baseline “no beaver” model exhibited only modest growth in woodland service value, the inclusion of expert-derived beaver effects resulted in significantly enhanced outcomes. The 50% (median) scenario showed sustained and substantial growth in ecosystem service value over time, aligning with expectations that beavers function as keystone species capable of delivering cumulative, landscape-scale benefits (Brazier et al., 2021; Puttock et al., 2017; Law et al., 2016).

The 5% quantile scenario, representing the lower bound of expert estimates, projected a decrease in NCAI values. However, this outcome should not be interpreted as a forecast of general decline under beaver reintroduction. Rather, it reflects a conservative worst-case scenario that incorporates the cumulative effects of minimal or negative impacts across all services. In contrast, the 95% quantile scenario, representing the upper bound of potential benefit, exhibited rapid growth in index values. Together, these bounds delineate a plausible range of outcomes, but the 50% median remains the most credible reflection of expected ecosystem service trajectories.

The application of SEJ and its integration into the NCAI framework also reveal several limitations. The current NCAI model does not include provisioning services, such as agricultural outputs or timber yield. This exclusion prevents a full accounting of potential trade-offs, such as the flooding of arable land or the loss of commercially valuable trees, impacts that have been widely noted via stakeholder engagement processes (Auster et al., 2020; Campbell-Palmer et al., 2016). These disservices, while likely to be spatially limited and context-dependent, remain invisible within the current national-scale framework. As a result, there is a risk of overestimating net gains without a more comprehensive and spatially explicit treatment of ecosystem service dynamics. Conceptual and methodological challenges also emerged in the process of translating expert knowledge into ecosystem service scores. For instance, the removal of the service category “mediation by biota” due to inconsistent expert interpretation underscores the need for greater definitional clarity and more

robust classification systems when applying SEJ in complex ecological contexts (Haines-Young & Potschin, 2018).

Despite these limitations, the implications for natural capital policy and NbS are clear. Beavers exemplify the potential for species reintroductions to generate multi-functional landscapes that support biodiversity, enhance climate resilience, and improve water quality. In this respect, they offer a cost-effective nature-based solution that aligns with international objectives on sustainable land management, ecosystem restoration, and climate adaptation (Albert et al., 2020; Puttock et al., 2025; Fairfax and Westbrook., 2025).

2.4.3 Performance of the Mixed Methods Approach

The integration of SEJ with the NCAI represents a novel, scalable, and policy-relevant method for assessing changes in ecosystem service provision in contexts where empirical data are limited. This mixed-methods approach combined expert-derived quantifications of service change with a nationally standardised index framework, enabling the incorporation of uncertainty into formal natural capital assessments for the first time. In doing so, it supports decision-making under uncertainty, a key challenge in rewilding, ecosystem restoration, and nature-based solutions (Bateman & Mace, 2020; Albert et al., 2020).

The online delivery of the SEJ protocol allowed for broad geographic participation and reduced logistical burden, securing input from experts across the UK and mainland Europe. However, the digital implementation also introduced challenges that are well recognised in expert elicitation studies (Knol et al., 2010). These included survey fatigue, occasional ambiguity in question phrasing, and linguistic barriers, issues which contributed to several experts withdrawing before completing the process. Overlapping or poorly defined terminology (e.g., confusion between “mediation by ecosystems” and “mediation by biota”) further hampered interpretation, leading to duplicated responses across distinct service categories. These findings reinforce the need for clearer conceptual framing, improved guidance materials, and potentially a modular survey structure that reduces cognitive load and facilitates service-specific engagement (Morgan, 2014).

A further limitation was the disciplinary skew within the expert panel, which predominantly comprised ecologists and natural scientists. As a result, cultural ecosystem services were poorly calibrated and ultimately excluded from the final decision-maker aggregation. This exclusion reflects broader challenges in quantifying qualitative and relational values using probabilistic methods (Chan

et al., 2012; Raymond et al., 2013). Future efforts should consider segmenting experts by service domain and developing tailored calibration sets that better align with disciplinary knowledge.

The choice of equal weighting over performance weighting emerged as a critical methodological decision in this study. In principle, performance weighting can yield more accurate pooled estimates (Cooke, 1991). However, in this application, performance weighting underperformed due to poor cross-domain calibration. The calibration set was necessarily broad, covering diverse service types to match the interdisciplinary scope of the assessment. Very few experts demonstrated strong calibration across all these domains, a common limitation in cross-disciplinary panels (Colson & Cooke, 2017).

In contrast, equal weighting distributed influence evenly across the panel, resulting in superior combined decision quality, measured as the product of calibration and informativeness. This choice likely contributed to the wider confidence intervals observed across ecosystem service scores, as it retained the full range of perspectives rather than discounting low-calibration contributions. While broader uncertainty is often viewed as a limitation, in this case it provides a transparent expression of genuine variability in expert opinion and real-world context dependency (Hemming et al., 2018). Under better-calibrated conditions, narrower uncertainty bounds would be expected, suggesting that future refinements in expert segmentation and training could restore the feasibility of performance-based aggregation.

Despite its limitations, the SEJ–NCAI integration represents a substantial methodological advancement. The use of quantile-based outputs (5%, 50%, and 95%) permitted the formal propagation of uncertainty through national-scale natural capital models. This level of transparency is rare in environmental accounting and offers a more nuanced foundation for policy decisions, particularly where rewilding strategies are concerned. It also supports a shift from deterministic assessments of ecosystem change to probabilistic and precautionary approaches that accommodate both potential gains and risks (Burgman et al., 2011; Polasky et al., 2015).

Looking forward, several improvements could enhance the utility and credibility of this mixed-methods framework. Segmenting expert panels by ecosystem service domain and using domain-specific calibration modules would improve performance weighting while maintaining fairness. Supporting resources such as handbooks, example questions, and training videos could reduce dropout rates and increase response quality. Additionally, integrating qualitative inputs from stakeholders, particularly land managers, Indigenous groups, and local communities, would allow for a fuller exploration of ecosystem service trade-offs and ground national-level assessments in local realities (Jacobs et al., 2016).

2.4.4 Future research

The findings of this study underscore both the potential and the complexity of using SEJ to evaluate how beaver activity shapes natural capital and ecosystem services. Wide uncertainty intervals highlight the need for further empirical validation, particularly for services mediated by indirect or context-dependent mechanisms. As beaver populations expand across Europe and North America, future research must clarify how these ecological changes affect the provision and resilience of natural capital at different spatial and temporal scales.

Water quality regulation emerged as a priority area. Experts emphasised the need for pollutant-specific studies, particularly on emerging contaminants such as pesticides and pharmaceuticals, which may behave unpredictably under fluctuating redox conditions in beaver wetlands. Temporal dynamics also warrant attention: the benefits and risks for water quality are likely to change with dam succession, from construction through to abandonment. Closely linked are gaps around chemical parameters such as pH, dissolved oxygen, and trace elements, which require site-specific monitoring to improve predictive capacity.

Flow and sediment regulation also require targeted research. Most evidence comes from small streams, leaving uncertainties about scaling to larger catchments and extreme weather events. Beavers clearly promote sediment retention and bank stabilisation, yet dam breaches or burrowing can trigger localised erosion (Larsen et al., 2020). Quantifying net sediment budgets over time and across geomorphic contexts is essential. Closely related is the need to better understand soil processes in post-abandonment meadows, particularly their capacity to act as carbon and nutrient sinks.

Biodiversity and habitat provision remain central to the beaver debate. While beaver ponds are consistently associated with high biodiversity (Law et al., 2019), potential trade-offs for disturbance-sensitive species remain underexplored. Research should move beyond descriptive habitat assessments to population- and community-level responses, including long-term impacts of coppicing, understory regeneration, and changes in species interactions.

Climate regulation represents another pressing knowledge gap. Beaver wetlands may sequester carbon in sediments and vegetation, but methane emissions remain poorly constrained. Longitudinal measurements of greenhouse gas fluxes under different hydrological and vegetative regimes are required to resolve the net climate balance. Microclimatic effects—such as canopy thinning and increased evapotranspiration, also deserve attention given their potential to buffer temperature extremes.

Finally, several service categories currently lack sufficient evidence to inform management. The role of beavers in pest and disease dynamics remains ambiguous, with plausible benefits through enhanced ecological resilience but also potential risks via vector habitat creation. Similarly, links to pollination and seed dispersal remain largely speculative. Both areas would benefit from hypothesis-driven, ecosystem-level studies that assess implications for agriculture and public health.

2.5 Conclusion

This study combined SEJ with the NCAI to evaluate how beavers influence woodland ecosystem services at a national scale. SEJ enabled a more comprehensive picture of ecosystem service benefits, trade-offs, and uncertainty, while NCAI translated these estimates into a formal valuation framework. Our findings show that beavers enhance the delivery of regulating services from woodlands. However, effects vary across services with varying levels of uncertainty underscoring the need for context-specific interpretation. The net effect on woodland assets was strongly positive, with exponential increases in natural capital value projected over time. This marks the first national-scale quantification of a species' contribution to ecosystem services in Scotland and offers a novel approach for incorporating uncertainty into environmental decision-making. The integrated SEJ-NCAI framework provides a scalable model for assessing rewilding interventions and informing nature-based policy.

3. Multifunctional farm wetlands and ponds: evidence for diverse ecosystem service provision

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Narrative author contribution statement

I led the design and development of this research, including the formulation of research questions and selection of the RAWES framework for ecosystem service assessment. I designed the field survey approach and conducted data collection across 60 RSuDS sites in central Scotland. I developed and implemented the spatial analysis and regression modelling to examine relationships between ecosystem service provision and landscape characteristics. I was responsible for data curation, validation, and the development of the analytical workflow and visualisations. I led the interpretation of the results, particularly in relation to multifunctionality and trade-offs, and wrote the manuscript, incorporating feedback from co-authors during revision.

Abstract

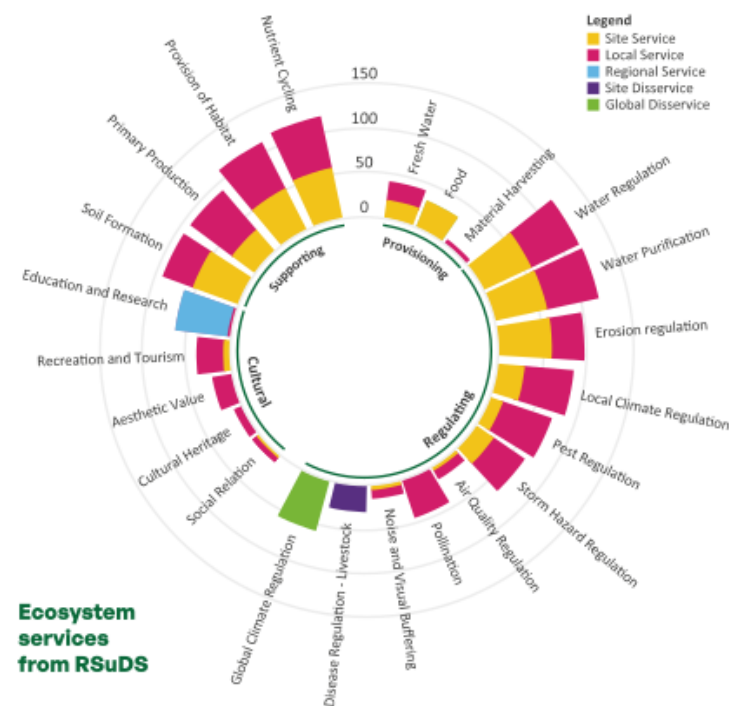
Nature-based solutions are increasingly promoted to address environmental challenges in agricultural landscapes, yet evidence of the ecosystem services they provide remains limited. This study evaluates the ecosystem services delivered by rural sustainable drainage systems (RSuDS), small ponds, wetlands, swales, and sediment traps designed to manage water quantity and quality in agricultural settings. Using the Rapid Assessment of Wetland Ecosystem Services (RAWES) framework, 60 RSuDS were surveyed across central Scotland to assess the diversity and distribution of ecosystem services.

Spatial analysis and regression modelling were used to examine associations between ecosystem service provision and landscape characteristics. Regulating services dominated, particularly those related to water quality improvement and flow regulation, while provisioning services such as food and water were less frequently recorded. Cultural services were also limited, partly reflecting restricted access and challenges in capturing intangible benefits. Supporting services formed an important foundation for wider ecosystem functioning.

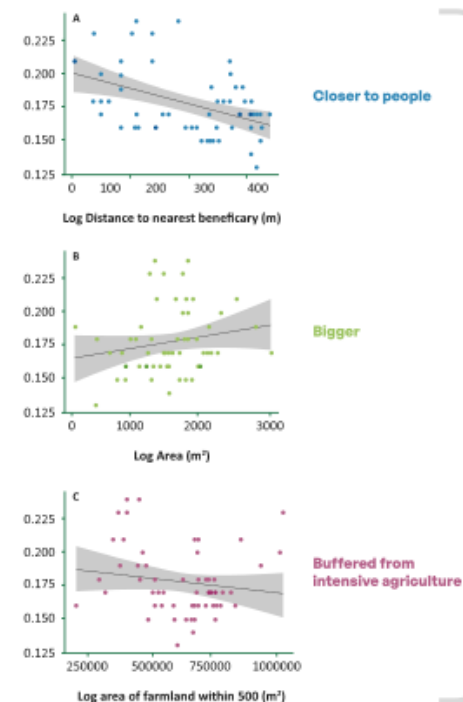
Ecosystem service provision was positively associated with RSuDS size and proximity to beneficiaries, while greater surrounding agriculture within a 500 m radius was negatively associated with service delivery. Potential disservices were also identified, particularly relating to greenhouse gas emissions and livestock disease risks. These findings highlight the multifunctional role of ponds and wetlands in agricultural landscapes and demonstrate how spatial configuration influences ecosystem service outcomes. The study provides empirical evidence to support improved assessment and place-based design of nature-based solutions that enhance environmental benefits while managing potential trade-offs.

Graphical abstract

Potential of wetlands and ponds to deliver payment for ecosystem services (PES)



Total ecosystem services from RSuDS



3.1 Introduction

A healthy environment underpins sustainable food production and food security (NFFN, 2023), yet intensive agriculture can degrade the natural capital that supports these outcomes. Natural capital comprises the renewable and non-renewable environmental assets that generate flows of benefits to people, known as ecosystem services (Costanza, 2020). Unsustainable agricultural practices can reduce these benefits by contributing to climate change (Laborde et al., 2021), biodiversity loss (Dudley & Alexander, 2017), soil degradation (Lal, 2013), and pollution of water bodies through point and diffuse nutrient runoff (Burkholder et al., 2006; Collins et al., 2018). Nutrient losses, particularly phosphorus and nitrogen, reduce drinking water quality (van den Brink et al., 2021), limit industrial and recreational water use (D’Arcy et al., 2022), and damage aquatic ecosystems (Kakade et al., 2021).

Despite these pressures, agricultural landscapes also present opportunities for ecosystem restoration. Approaches such as agroecological intensification and regenerative agriculture demonstrate that food production and ecosystem recovery can coexist (Pretty et al., 2018; Lal, 2020; Giller et al., 2021). Given that 38%, 71%, and 80% of land in the European Union, United Kingdom, and Scotland respectively is used for agriculture (Eurostat, 2022; National Food Strategy Independent Review, 2021; Brand et al., 2021), improving environmental management within farmland has substantial potential to support nature recovery at landscape scales.

Agricultural policy increasingly reflects this shift. In the EU and the UK, reforms to the Common Agricultural Policy (CAP), Environmental Land Management Schemes (ELMS), and Scotland’s Agri-Environment Climate Scheme (AECS) aim to support farming systems that deliver environmental benefits alongside food production (Pe’er et al., 2019). Achieving these outcomes requires effective interventions that enhance ecosystem services within managed landscapes.

Nature-based solutions (NbS) address societal challenges by working with natural processes to generate environmental, social, and economic benefits while enhancing ecosystem resilience (Seddon et al., 2020). Within agricultural systems, NbS such as ponds, wetlands, and vegetated landscape features can regulate water flows, improve water quality, and support biodiversity while contributing to wider ecosystem service delivery (Xu et al., 2020).

One such intervention is Rural Sustainable Drainage Systems (RSuDS), which include ponds, wetlands, swales, sediment traps, and bunds designed to intercept and treat agricultural runoff before it enters downstream watercourses (Duffy et al., 2016; Blackwell and Pilgrim, 2011). Unlike conventional drainage systems that rapidly remove water through pipes and ditches, RSuDS slow, store, and treat water within the landscape (Everard et al., 2017; Thorslund et al., 2017). Their

primary or “anchor” service is water quality and quantity regulation, the key function for which they are typically implemented. Anchor services are the main ecosystem services motivating interventions, but they often enable additional co-benefits across other ecosystem service categories.

The effectiveness of RSuDS arises from interacting physical, chemical, and biological processes. Reduced flow allows sediments and associated nutrients to settle, while longer water residence times promote nutrient transformation through processes such as denitrification and phosphorus sorption (Vymazal, 2007; Tanner et al., 2012). Vegetated margins and shallow water zones increase contact between water, soils, and plant communities, supporting nutrient uptake, organic matter processing, and habitat provision (Kadlec and Wallace, 2009; Mitsch and Gosselink, 2015). Together, these processes reduce nutrient export, attenuate peak flows, and support a broader range of regulating and supporting ecosystem services (Mitsch et al., 2015; Xu et al., 2020).

Despite these benefits, decision-making often focuses narrowly on anchor services, while the wider ecosystem service co-benefits of NbS remain under-recognised (Everard, 2014; Gouriveau, 2009). This bias partly reflects the relative ease of measuring services such as water regulation or carbon cycling compared with more complex social or ecological benefits (Kadykalo and Findlay, 2016; Villa and Bernal, 2018). In addition, the absence of widely adopted assessment frameworks can make it difficult to evaluate ecosystem service outcomes consistently across sites (Mitsch et al., 2015; Xu et al., 2020).

As RSuDS are increasingly promoted within Scotland’s agricultural reforms to address climate change, restore nature, and support food production, understanding their full range of ecosystem services is essential. While not all services can be directly quantified, the Rapid Assessment of Wetland Ecosystem Services (RAWES) framework provides a standardised approach for evaluating their relative significance through scores. By integrating qualitative and quantitative information, RAWES enables a more holistic assessment of ecosystem service delivery and highlights both benefits and potential trade-offs across landscapes.

This study advances the evidence base for RSuDS within the ecosystem services framework. Its aims are to: (1) assess the ecosystem services provided by RSuDS, using indicators to capture their relative significance across a broad range of benefits; (2) determine the spatial and demographic factors influencing variability in these services; and (3) propose place-based design enhancements to optimise benefits and minimise trade-offs. Although the study is situated in lowland agricultural landscapes in central Scotland, the processes examined, runoff interception, water retention, nutrient transformation, and spatial relationships between wetlands, land use, and beneficiaries, are

common to intensively managed agricultural systems globally. As such, the findings are relevant to a wide range of temperate agricultural regions facing similar challenges related to diffuse pollution, flood risk, biodiversity loss, and the need to multifunctionally manage agricultural land.

3.2 Methods

3.2.1 Study areas and data collection

In total, 60 rural sustainable drainage systems (RSuDS) were surveyed in central Scotland during June–August 2022 (Figure 1). The study focused on lowland agricultural landscapes (<200 m elevation), which represent the most productive agricultural areas in Scotland and where diffuse agricultural pollution pressures are high. The surveyed RSuDS comprised a range of feature types, including ponds, wetlands, sediment traps, swales, and small bunded water features, either constructed or semi-natural, that were designed or functioning to intercept agricultural runoff.

Potential RSuDS were identified using a combination of Ordnance Survey maps, high-resolution satellite imagery (Google Earth), and local knowledge, with features selected where they were hydrologically connected to agricultural land and likely to receive surface or subsurface runoff. Site selection was purposive rather than random, with the objective of capturing a wide diversity of RSuDS types, sizes, surrounding land uses, and degrees of proximity to potential beneficiaries.

Final inclusion of sites was constrained by practical considerations, including landowner permission, site accessibility, and safety for field assessment. Features that were permanently dry, heavily engineered with no biological component, or clearly disconnected from agricultural drainage pathways were excluded. The resulting set of 60 RSuDS represents a heterogeneous sample spanning intensive arable systems to low-density livestock grazing, suitable for exploring how ecosystem service delivery varies with landscape and demographic context.

3.2.2 Ecosystem service assessment

3.2.2 Ecosystem service assessment

A field-based approach, informed by the updated ‘Rapid Assessment of Wetland Ecosystem Services’ (RAWES), was used to evaluate the ecosystem services provided by RSuDS. RAWES follows a defined assessment protocol as developed and outlined in previous research (McInnes & Everard, 2020; McInnes & Everard, 2017; Ramsar, 2017). RAWES takes a semiquantitative approach, integrating evidence from observations and expert knowledge across the ecosystem services spectrum. Unlike

traditional assessments, RAWES does not assign quantitative or monetary values to ecosystem services. Although specifically developed for wetland assessment, RAWES can be applied to a range of habitat types and can be used across a range of scales from whole landscapes to localised zones of interest (McInnes and Everard, 2017). It assesses four categories of wetland ecosystem services, supporting (ecological processes that underpin ecosystem functioning), provisioning (material goods such as food and water), regulating (benefits derived from the regulation of ecological processes, including water quality and flow), and cultural (non-material benefits such as recreation, education, and aesthetic value), adapted from the Millennium Ecosystem Assessment (Corvalán and Reid, 2001) (Table 5).

Table 5: Potential ecosystem services associated with wetlands and pond ecosystems (Corvalán and Reid, 2001)

Ecosystem Service	Description
Provisioning	
Fresh water available for abstraction and use	Availability of water for various human and ecological uses.
Food production	Cultivation or harvesting of food sources like crops, fruit, and fish.
Fibre and fuel production	Harvesting resources like timber, wool, and other materials for fuel and fiber.
Genetic resources	Use of genetic material for crop or livestock breeding and biotechnology.
Biochemicals, natural medicines, pharmaceuticals	Derivation of natural substances for medicine and chemical applications.
Ornamental resources	Decorative natural items like shells and flowers.
Harvesting of clay, minerals, aggregates, etc.	Extraction of geologic resources for construction, manufacturing, and other uses.
Waste disposal	Use of natural systems for processing and assimilating waste.
Energy harvesting from air and water flows	Utilization of wind and water flows for renewable energy generation.
Regulating	
Air quality regulation	Natural systems filtering pollutants and maintaining air quality.
Local climate regulation	Regulation of microclimates, temperature, and precipitation at local scales.
Global climate regulation	Greenhouse gas sequestration and influence on global temperature and weather patterns.
Water regulation	Management of runoff, flooding, and water flow timing and scale.
Natural hazard regulation	Ecosystems providing protection from storms, floods, and similar hazards.
Pest regulation	Control of pests that affect crops or natural systems.
Disease regulation - human	Mitigation of disease transmission and prevalence in human populations.
Disease regulation - stock	Reduction in the spread of diseases among livestock.
Erosion regulation	Prevention of soil erosion and maintenance of land stability.
Water purification and waste treatment	Natural cleansing of water and decomposition of waste.
Pollination	Support for pollination of plants, crucial for agriculture and ecosystem health.
Salinity regulation	Prevention or reduction of soil salinity buildup.
Fire regulation	Influence on the likelihood and severity of wildfires in the ecosystem.
Noise and visual buffering	Reduction of noise pollution and enhancement of visual landscapes.
Cultural	
Cultural heritage	Preservation and enhancement of areas with historical or cultural importance.
Recreation and tourism	Provision of natural spaces for leisure, tourism, and outdoor activities.
Aesthetic value	Enjoyment derived from the visual beauty of ecosystems.
Spiritual and religious value	Ecosystems supporting spiritual practices and religious connections.
Inspiration of art, folklore, architecture, etc.	Natural settings inspiring creative works and cultural practices.
Social relations	Ecosystems facilitating community activities like fishing, grazing, or farming.
Educational and research	Opportunities for learning and scientific exploration in natural environments.
Supporting	
Soil formation	Development and maintenance of fertile soils.
Primary production	Creation of biomass through photosynthesis as the foundation of food webs.
Nutrient cycling	Recycling of nutrients essential for ecosystem productivity.
Water recycling	Natural recycling of water within ecosystems.
Photosynthesis	Production of oxygen and energy through the capture of sunlight by plants.
Provision of habitat	Support and maintenance of habitats for diverse species.

Within each ecosystem service category, there are specific services of relevance (Table 5). These are assigned a value from -1 to +1 in 0.5 intervals. Positive scores indicate a benefit (service) to society, while negative scores indicate a disservice. The difference between a score of 0.5 and 1 reflects whether the benefit or disservice is considered minor or major. This distinction is context-dependent and should be defined by assessors prior to evaluation. In this study, the minor/major threshold was based on the estimated number of people affected by the service, with a major service defined as one benefiting more than 100 people. This threshold is commonly used in low-density areas (RRC-EA, 2020), with rural Scotland averaging 67 people per square kilometre (Scottish Government, 2021). Importantly, ecosystem services are only considered to exist when a clear benefit to identifiable beneficiaries can be established; otherwise, they are classified as potential rather than actual services. Each ecosystem service can operate at different spatial scales, which are defined by the surveyor (Table 6), and may span multiple scales. RAWES assessments are conducted at a single point in time, and do not incorporate temporal variability.

Table 6: Spatial scale of service provision categories used in the assessment of Scottish RSuDS with justification.

Scale category	Description	Justification
Site	Within the delineated boundary of the RSuDS	Physical processes within the RSuDS
Local	Outside the delineated boundary but within the farm/local catchment boundary	RSuDS are located within a specific farming community and the wider catchment
National	Outside of the farm/local catchment but within the United Kingdom	Services can spill over from the farm/field into other parts of the country e.g., education
Global	Globally, but outside of the United Kingdom	Services can occur beyond national borders e.g., global climate regulation

The scores are then transformed into semiquantitative Ecosystem Service Index (ESI) values, indicating overall outcomes. These ESI values can be applied to service categories (regulating, provisioning etc.) and the 4 service categories can also be combined to create an ESI that represents the total value from specific RSuDS (Cianchi et al., 2023).

3.2.3 Place based assessment

To examine how spatial and demographic characteristics influence ecosystem service provision, RSuDS were analysed in QGIS (QGIS.org, 2023) using data derived from drone mapping and satellite imagery. Drone surveys were conducted during field visits in summer 2022 using a DJI Phantom 4 Pro unmanned aerial vehicle, capturing high-resolution aerial images of each RSuDS and its immediate surroundings. These images were used to delineate feature boundaries, calculate RSuDS area, and identify prominent landscape features such as field margins, access tracks, roads, and footpaths.

Satellite imagery was obtained from Google Earth and accessed contemporaneously with field surveying in 2022. This imagery was used to measure the distance from each RSuDS to the nearest identifiable beneficiary of ecosystem services (e.g. residential buildings, footpaths, roads, or community features), providing a consistent spatial reference for distance-based analyses (Figure 8).

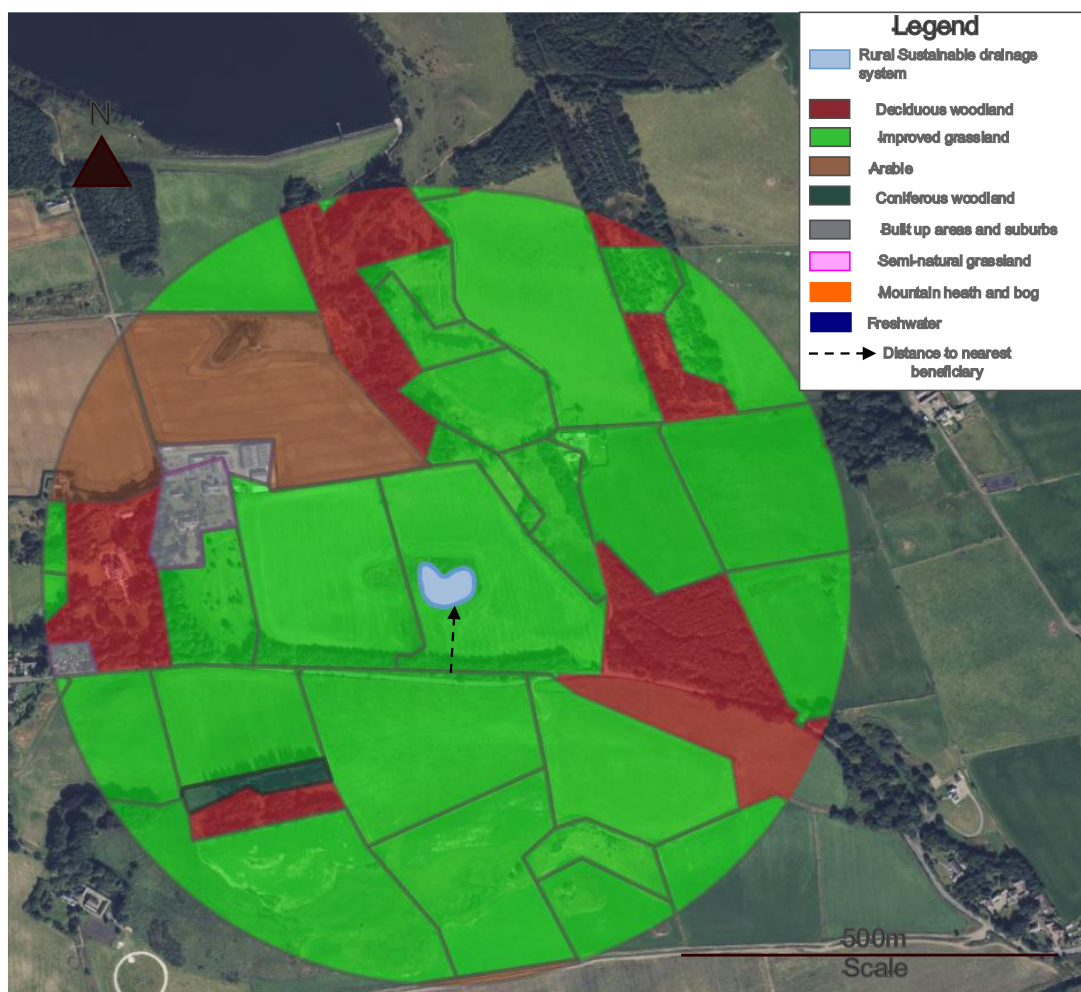


Figure 8: RSuDS with a 500m buffer comprising of several types of land use. The black line shows the distance to the nearest beneficiary which in this case was a road.

Land use was characterised for the surrounding area within a 500m radius from the RSuDS. This radius distance was chosen to encapsulate the different scales at which RAWES operates (Table 6) and because it has been successfully used in terrestrial (Fuentes-Montemayor et al., 2012) and aquatic ecology (Law et al., 2019) land use studies. The buffer was used to extract land use type from the UKCEH Land cover map 2021 (Marston et al., 2022). The UKCEH Aggregate class land use dataset divides land use into 10 categories. Within each RSuDS buffer, land cover was expressed in m².

Spatial and demographic variables were analysed using regression modelling to examine their statistical associations with ecosystem service values. Our dependent variable was the total ESI of the 4 ecosystem service categories measured for each RSuDS. Several independent variables were considered for the regression modelling but due to constraints with the dataset not all were taken forward to the final model (Table 7). All statistical analysis and regression modelling was conducted using R (R Core Team, 2021). Prior to regression modelling, the distance to nearest beneficiary and the area of the RSuDS were log transformed to reduce the effect of outliers. All continuous explanatory variables were mean centred and scaled to enable comparison on a standardised scale. Correlations between explanatory variables were assessed in a correlation matrix and checked for variance inflation with no further correlations observed. Two sites were removed as outliers that had a disproportionately large effect on the model because they were unrepresentative of the wider dataset (i.e., very large wetlands and very different surrounding land uses).

Table 7: Showing the different independent variables considered for regression modelling, the final variables chosen for the modelling and explanations for the choices.

Independent variable	Used in final model	Explanation
Coniferous woodland	N	Not included because this land-use type was absent (0 m ²) within the 500 m buffer for more than half of the RSuDS sites.
Semi-natural grassland	N	Not included because this land-use type was absent (0 m ²) within the 500 m buffer for more than half of the RSuDS sites.
Mountain, heath and bog	N	Not included because this land-use type was absent (0 m ²) within the 500 m buffer for more than half of the RSuDS sites.
Deciduous woodland	Y	Included because the relationship between woodland and water at a catchment scale (Baulenas et al., 2021)
Improved grassland	Y	Combined into a single composite called 'farmland' due to positive correlations. Included because of the effect agriculture has on aquatic ecosystems (Basso et al., 2020)
Arable	Y	
Freshwater	Y	Included because of the interconnected nature of aquatic ecosystems (Sayer, 2014).
Built up areas and suburbs	Y	Included because RAWES assesses ecosystem services in relation to nearby populations
Distance to nearest beneficiary	Y	
Area of wetland	Y	Included because of the relationship between wetland size and ecosystem service valuation (Zhou and Gong ,2020).

All possible iterations of the model were generated using the glmulti package (Calcagno and Mazancourt, 2010) to find the best fitting model for Akaike information criterion corrected for small sample sizes (AICc) and to reduce model complexity. Interactions between the explanatory variables were tested and none were found. The residuals of the generalised linear model (GLM) exhibited a non-normal distribution with a positive skew and a long posterior tail. This made the best fitting model a GLM incorporating a gamma distribution and log link function.

Each RSuDS contributed a single observation to the dataset, and therefore site identity was not suitable as a random effect. Although sites were distributed across central Scotland, all were located within broadly similar lowland agricultural landscapes. To assess whether broad-scale spatial clustering influenced model outcomes, site coordinates were initially explored as a potential grouping structure; however, no meaningful improvement in model fit was observed, and inclusion of a spatial grouping term increased AICc. As a result, no random effects were retained, and final analyses were conducted using generalised linear models. The same regression modelling process was applied to the individual ecosystem service categories (regulating, cultural, provisioning, and supporting). Consistent effects were observed for supporting and regulating services; however, provisioning and cultural services could not be robustly modelled because the number of services recorded within these categories was low and exhibited limited variation among sites, resulting in insufficient explanatory power for statistical analysis.

3.3. Results

3.3.1 *Diversity of ecosystem services*

The highest number (623 counts) and diversity (9 subcategories) of ecosystem services were found in the regulating category (Figure 9). Regulating services associated with water, with respect to either quantity or quality, form 38% of the regulating services provided. In total, 18% of the regulating services were related to insects that provide pest regulation and pollination. Local climate regulation formed 13% of the services whilst erosion regulation accounted for 15% of the services delivered.

Supporting services was the second largest category (415 counts over 4 subcategories), but this is to be expected as they are the foundation from which the other services are derived. This follows the same trend in scale of services as total and regulating services and, similar to regulating services, most of the supporting services are minor (Figure 9).

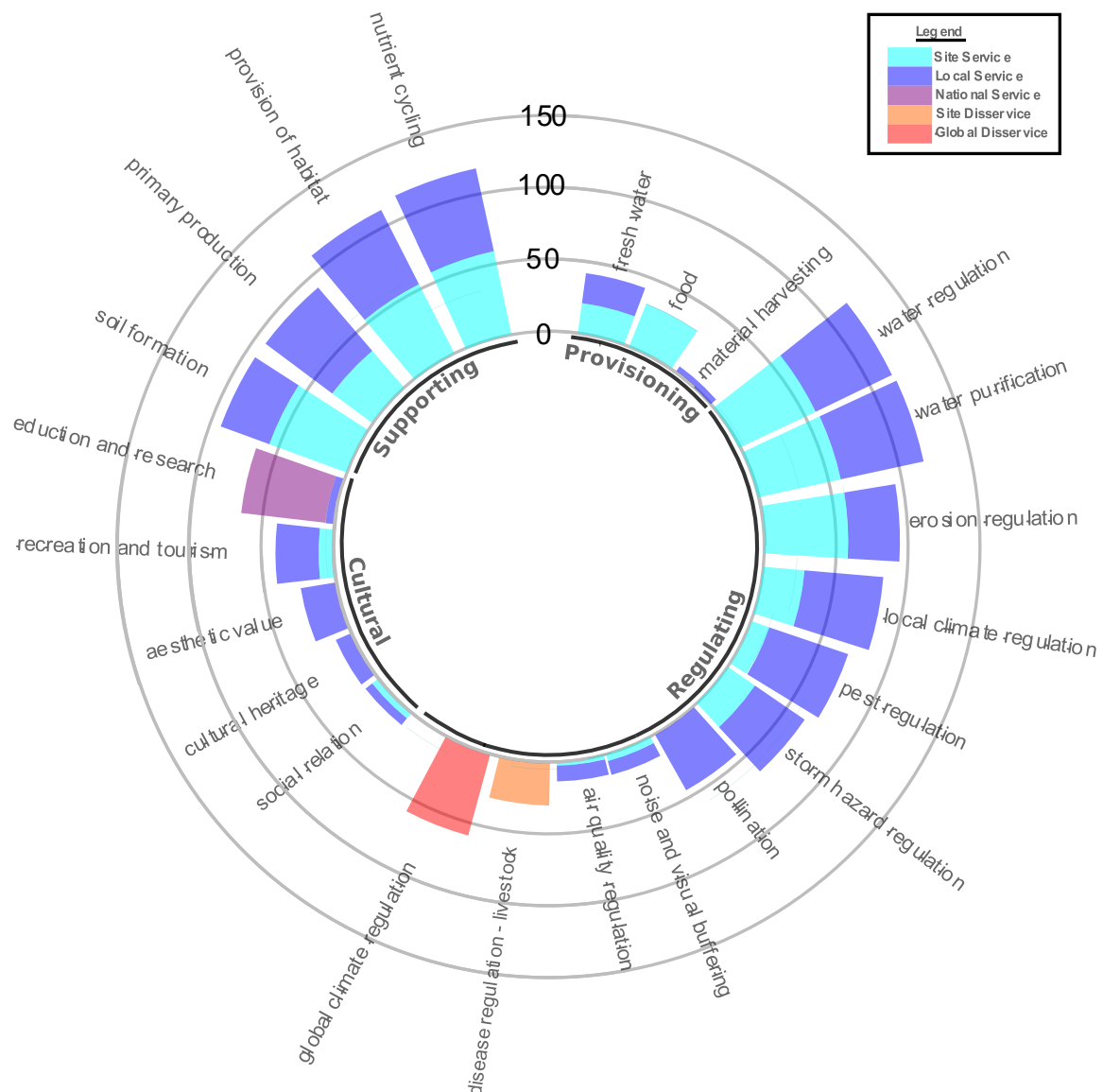


Figure 9: Ecosystem services and disservices from 60 RSuDS by category and scale (site, local, national, global). These count data do not differentiate between minor and major services.

Both disservices of RSuDS are found within the regulating category (Figure 9). Disease regulation of livestock (17 disservice counts) all occurred at a site scale i.e. livestock liver fluke. These are all deemed minor disservices as they affect a small farming community. Global climate regulation relates to the role of RSuDS in carbon cycling where 60 counts of disservices to society were recorded. Global climate regulation relates to the role of RSuDS in carbon cycling, where 60 counts of disservices to society were recorded. These are classified as minor disservices, as the magnitude of impact from individual ponds is low, despite the global scale of the issue. No disservices at a national scale were recorded.

Cultural services (154 counts over 5 subcategories) are dominated by education and research (42%), which highlights that RSuDS are a valuable source of learning (Figure 9). The second largest service within this category was recreation and tourism (26%), relating to footpaths, fishing and duck hunting. The scale of delivery for these services differs from the others in that, relative to other services, it has a significant national impact (38%), but limited site-scale significance (10%). This is due to the large scale of influence of education and research, which has national significance. The cultural services provided were primarily deemed minor. The smallest amount (79 counts) and range (3 subcategories) are found in provisioning ecosystem services, primarily as fodder and water for livestock. Overall, 8 of the 60 RSuDS surveyed had some form of mineral harvesting occurring.

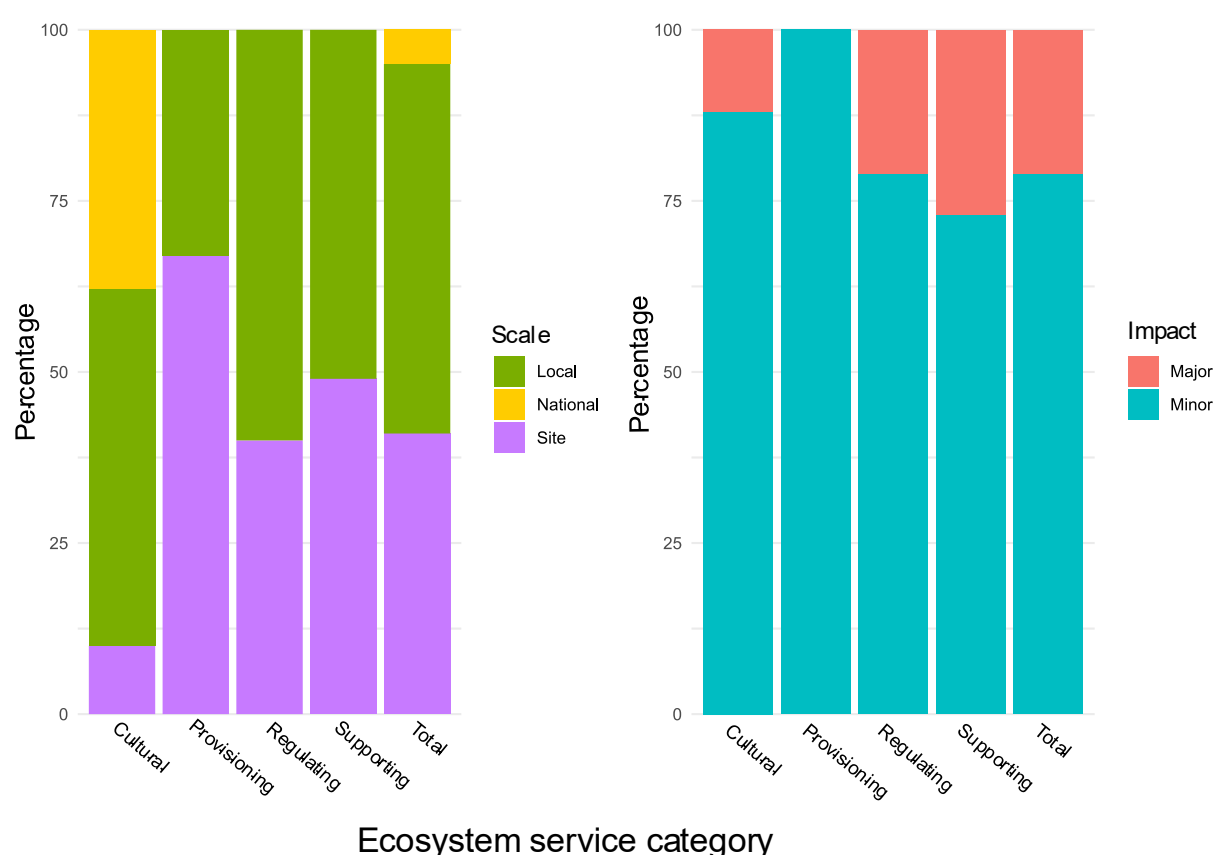


Figure 10: Breakdown of total services for RSuDS and each service category by scale. It also shows the percentage for both minor services and major services for each category.

The majority of total services are classed at site and local level with a small proportion at national and none at global (Figure 10), though we did find global climate regulation to be a disservice at global scale. More than 2/3 of the regulating services were minor (Figure 10). The spatial scale of regulating service delivery followed the same pattern as the total services, but there were no national regulating services. Most provisioning services are delivered at site and then local scale with none at regional or global. All the provisioning services were minor (Figure 10).

3.3.2 Drivers of ecosystem services

Three variables were significantly associated with variation in total ecosystem services provided by RSuDS (Figure 11). The proportion of farmland within the 500 m radius surrounding each RSuDS was negatively associated with total ecosystem service values, such that sites embedded in more intensive agricultural landscapes tended to deliver fewer services. In contrast, RSuDS area was positively associated with total ecosystem services. Distance to the nearest beneficiary was also negatively associated with ecosystem service provision, with more distant RSuDS exhibiting lower total service values. All three explanatory variables had similar effect sizes within the model, suggesting comparable relative importance in explaining observed variation

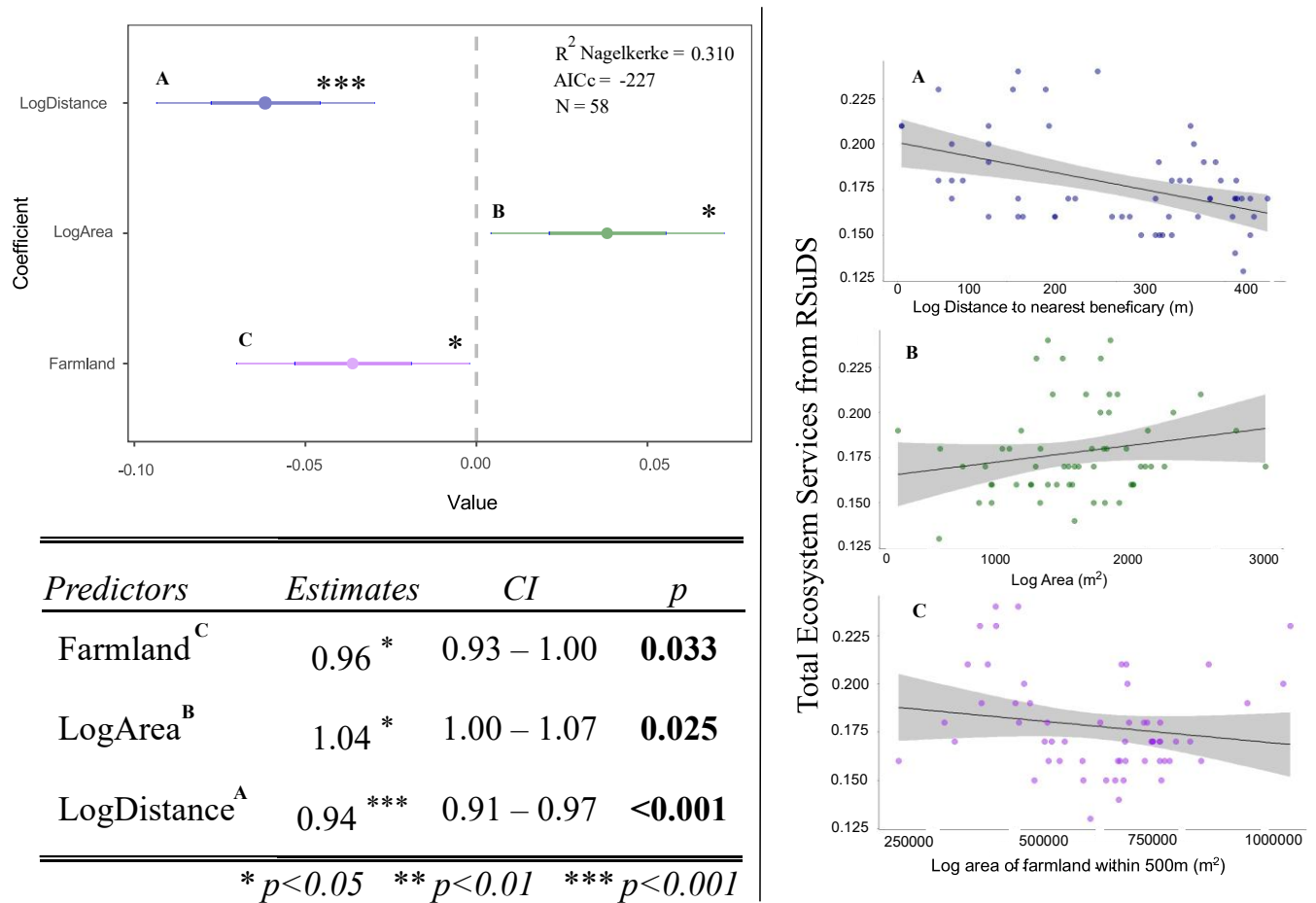


Figure 11: Outputs of the GLM. The caterpillar plot in the top left shows how each of the coefficients affects total ecosystem services. The 3 scatter graphs on the right give more detail on how each of the coefficients interact with total ecosystem services. The table in the bottom left gives more details for each coefficient with estimates, confidence intervals and p values.

3.4. Discussion

This study assessed the diversity of ecosystem services delivered by RSuDS, providing new empirical evidence of how these services are generated and distributed across spatial scales in agricultural landscapes. By identifying spatial and demographic factors associated with variation in ecosystem service provision, the results highlight how the design and location of ponds and wetlands can influence the benefits they provide and the trade-offs that may arise.

3.4.1 Services provided by RSuDS

The high number of regulating services delivered by RSuDS is consistent with findings from other wetland ecosystem service assessments, where regulating functions typically dominate over provisioning and cultural services (De Groot et al., 2012; RAMSAR., 2018). Similar patterns have been reported in studies applying ecosystem service assessment frameworks to wetlands and small freshwater systems in both Europe and elsewhere, where services related to water quality regulation, flood mitigation, and nutrient retention frequently emerge as the most prominent benefits (McInnes and Everard, 2017; Cianchi et al., 2023). Furthermore, like other wetland habitats, regulating services associated with RSuDS are largely linked to water quality and hydrological regulation (Xu et al., 2020). This uneven consideration of ecosystem services can lead to designs that prioritise a narrow set of objectives, most notably water quality improvement (Gouriveau, 2009). Such services are often favoured because their benefits are tangible, visible, and readily attributable to specific interventions, making the link between action and outcome clear to local beneficiaries and decision-makers across multiple spatial scales. For example, landowners who create wetland features on their land can directly observe reductions in local flood risk and soil erosion, reinforcing the emphasis on these regulating services in design and management decisions (Acreman and Holden, 2013). In addition, benefits transfer to downstream communities, which can be inferred from mapped data of downstream communities likely to benefit from having upstream flood storage and pollutant removal (Blackwell & Pilgrim, 2011). However, it is important to acknowledge the potential issue of scale-related bias, as benefits at the local level are often more immediately apparent and measurable, while broader cumulative benefits at regional or global scales may be less readily captured in ecosystem service assessments (Lindborg et al., 2017).

Cultural service provision was dominated by education and research, highlighting the opportunities for interaction, learning, and study afforded by RSuDS. This aligns with a key research gap identified by Hill et al. (2021), with this study demonstrating how the educational value of RSuDS can contribute to the health and well-being of rural communities. Recreation and tourism are also important cultural services associated with wetlands, as noted in other studies (Cuenca-Cambronero

et al., 2023). However, for these services to provide actual benefits rather than unrealised potential, the wetland must be physically and socially accessible to those engaging in recreation or tourism activities. Overall, cultural services were infrequent, likely due to the difficulty of incorporating the perspectives and knowledge of local communities and stakeholders (Plieninger et al., 2013) but also because most of the RSuDS were on private land which limits access to local people in an area of low population density. These combined factors resulted in low amenity value for the local community. As a result, the assessment of cultural services is likely to be an underestimation, reflecting a common theme when evaluating ecosystem services in aquatic and rural landscapes (Rodrigues, 2015; Norton et al., 2012). This is likely because cultural services are often intangible and do not leave an observable evidence-trail for the surveyor (Fish et al., 2016; Xu et al., 2020). Additionally, interaction with, and monitoring of, cultural services can be temporally sporadic, as demonstrated for health outcomes linked to exposure to blue space environments (Grace et al., 2024).

Assessments may also be affected by the timing of surveys not aligning with key periods of benefit delivery leading to an incomplete picture of the full cultural value provided by RSuDS. Furthermore, cultural services often depend on unique landscape features or cultural narratives that resist generalisation, and they vary temporally and among stakeholders, which undermines attempts at consistent measurement (Tew et al., 2019; McElwee, 2022).

Future research should incorporate more formalised community engagement and mixed-method approaches to improve the identification and representation of cultural ecosystem services.

Participatory methods such as workshops, interviews, and participatory mapping can integrate local knowledge and stakeholder perspectives, helping to capture shared and contested cultural values that are often intangible or relational (Daniel et al., 2012; Plieninger et al., 2013; Gould et al., 2019; Hubatová et al., 2023; Khirfan et al., 2021). broaden representation and reduce participation barriers, studies may triangulate in-person methods, such as participatory mapping, questionnaires, and interviews, with digital data sources including social media images and text, which can provide additional proxies for cultural ecosystem service use and valued landscape attributes (Romanazzi et al., 2023; Fox et al., 2021; Huai et al., 2022). The design and implementation of such approaches was beyond the scope of this study but represents an important direction for future work aimed at strengthening the evidence base for cultural ecosystem services (Grace et al., 2023).

Food and water were the two main provisioning services recorded, consistent with findings from the broader wetland ecosystem services literature across a range of geographic contexts and wetland types; however, provisioning services remain comparatively understudied relative to regulating and supporting services (Xu et al., 2020). This may be due to underutilisation of RSuDS within the farming community because of their low perceived provisioning value (Eric et al., 2022). It is straightforward

for a surveyor to connect foundational processes to many supporting services and this likely explains the high number identified in our study. For example, water purification would not occur in the absence of soil formation and primary production to enable vegetation and bacteria to mitigate pollutants. Habitat provisioning and nutrient cycling are the two largest service categories found in this study, again consistent with findings of other research (Xu et al., 2020).

3.4.2 Global climate regulation disservice

In this study, RSuDS were conservatively treated as potential net emitters of greenhouse gases, based on evidence from analogous small freshwater ecosystems. Research from ponds, wetlands, swales, ditches, and stormwater basins indicates that these systems can exhibit both high carbon burial rates (Fennessy et al., 2018; Ray & Holgerson, 2023; Rogers et al., 2022; Euliss et al., 2006; Jeffries et al., 2022) and substantial greenhouse gas emissions (Ray et al., 2023; Holgerson et al., 2023; Gilbert et al., 2021; Holgerson & Raymond, 2016; Peacock et al., 2021; Rabaey & Cotner, 2022). However, studies that explicitly combine burial rates with greenhouse gas flux measurements to determine net carbon balance remain limited for many wetland systems (Goeckner et al., 2022; Ljung & Lin, 2023; Taylor et al., 2019) and are largely absent for RSuDS. As a result, the overall sink–source status of these systems remains uncertain. Carbon dynamics in small waterbodies are known to vary considerably depending on factors such as nutrient enrichment, sedimentation rates, hydrological regime, and oxygen availability (Mitsch and Gosselink, 2015; Peacock et al., 2019). While wetlands can act as net carbon sinks in some circumstances (Hambäck et al., 2023), RSuDS located in intensively managed agricultural landscapes may experience elevated greenhouse gas emissions due to high nutrient inputs and physical disturbance (Peacock et al., 2019; Malerba et al., 2022).

Indicators of the RSuDS' overall 'health' relating to greenhouse gas emissions could be incorporated into future RAWES guidance through simple, cost-effective metrics such as vegetation cover, water level fluctuations, and bare earth exposure, which can act as proxies for underlying biogeochemical processes (Reddy et al., 2022). Quantifying these uncertainties is important for global greenhouse gas budgets (Lauerwald et al., 2023) but given the inherent variability of natural and managed ecosystems, pragmatic approaches often rely on conservative assumptions and buffers rather than exhaustive uncertainty reduction (Maron et al., 2018).

While sequestering carbon is a critical component of addressing climate change, it is important that it does not dominate decision-making at the expense of a more holistic perspective. RSuDS offer a wide range of ecosystem services beyond carbon sequestration, many of which have been lost with the decline of ponds and wetlands in agricultural landscapes (Alderton et al., 2017). These include

services critical for climate adaptation, such as hydrological buffering, support for natural pest predators and pollinators, and habitat provision for biodiversity. Furthermore, other Nature-based Solutions and land use strategies can also contribute to carbon sequestration (Seddon et al., 2020). A balanced perspective ensures we can address multiple environmental challenges, climate mitigation, adaptation, biodiversity loss, and water quality, simultaneously, rather than focusing narrowly on emissions alone.

3.4.3 Ecosystem service trade-offs at different spatial scales

To minimise trade-offs in ecosystem services, we first need to understand how, where and when they occur. For RSuDS, the primary trade-offs relate to access. This occurs in two ways; the first is livestock having access to freshwater and fodder from the RSuDS. This results in disturbance of sediment, which in turn can release greenhouse gases (Malerba et al., 2022a). Furthermore, grazing in these wet areas increases the risk of liver fluke (Innocent et al., 2017; Olsen et al., 2015). Allowing livestock access to RSuDS for fodder and water will prioritise provisioning services at the expense of regulating disservices. This trade-off trend is repeated with mineral harvesting from the RSuDS, whereby sediment is extracted and reapplied to agricultural land to enhance soil conditions. Whilst this was only recorded in a few instances in our study, the disturbance creates a global climate regulation disservice (Fennessy et al., 2018). However, current RSuDS guidance encourages the removal of mineral material to help promote effective functioning of RSuDS (Duffy et al., 2016).

The second issue relates to public access. Cultural services such as recreation, education, and social value are derived from RSuDS when people can visit and interact with these environments (Hambäck et al., 2023). However, increased visitation of these areas may result in increased sediment disturbance which affects the ability of the RSuDS ability to sequester carbon and therefore, the global climate regulation service (Malerba et al., 2022b). In this scenario, cultural services are increased but also increase the disservice of global climate regulation.

Such trade-offs will occur at different spatial and temporal scales (Thorslund et al., 2017) and need to be explicitly considered when assessing for equitable ecosystem service delivery across different communities. For example, landowners and local communities benefit largely at the site and local scale from provisioning and cultural ecosystem services, whilst a disservice of climate regulation occurs at a global scale. This 'scattering' of ecosystem services over several beneficiaries (Toxopeus & Polzin, 2021) means it is essential to engage with stakeholders to determine how trade-offs might impact specific communities and to understand the potential 'winners' and 'losers' (Guida et al., 2016).

Navigating trade-offs among ecosystem services requires transparent and inclusive decision-making processes that consider both benefits and potential disservices (Howe et al., 2014; Pascual et al., 2017). Participatory approaches, such as scenario planning, can support stakeholders in exploring alternative management strategies, visualising potential outcomes, and identifying acceptable compromises (Enengel et al., 2011). In agricultural landscapes, practical management measures can help balance ecosystem service provision. For example, restricting livestock access through rotational grazing or exclusion zones with alternative water provision can reduce disturbance while maintaining provisioning benefits, whereas designated pathways, boardwalks, or zoning can allow public access while minimising sediment disturbance and ecological impacts (Newsome et al., 2013).

Adaptive management is also important, as ecosystem service trade-offs may shift over time with changing environmental conditions and land-use practices. Regular monitoring and periodic review of management approaches can help ensure that interventions continue to deliver desired outcomes while minimising unintended consequences (Farley and Costanza, 2010). Effective governance therefore requires collaboration among land managers, policymakers, and local communities to ensure that decisions reflect both ecological processes and social priorities (Xu et al., 2018). Placing communities at the centre of decision-making can strengthen local stewardship, build trust, and support the long-term sustainability of nature-based interventions (Brownson et al., 2019; Rodgers and Kendall, 2023).

3.4.4 Place-based design to optimise RSuDS benefits

Understanding the social and demographic context in which nature-based solutions are implemented can help inform the spatial design and placement of features such as ponds and wetlands within agricultural landscapes (Mokondoko et al., 2018). Our results suggest that RSuDS located closer to people tend to deliver greater observable ecosystem service benefits, likely because proximity increases opportunities for interaction with and recognition of these services (Hambäck et al., 2023; Eric et al., 2022). This relationship is also reflected in the RAWES framework, where proximity to beneficiaries is considered when evaluating certain ecosystem services, particularly cultural services (RRC-EA, 2020). Similar patterns have been observed in other wetland studies, where ecosystem service benefits are more readily realised in areas with higher population density or greater accessibility (Johansson et al., 2019; Pedersen et al., 2019; Eric et al., 2022; Brander et al., 2013). However, most RSuDS examined in this study were located in low-density rural landscapes with limited public access, which may partly explain the relatively low cultural service

values recorded. Enhancing access where appropriate could therefore increase the cultural benefits associated with these systems.

The positive association between RSuDS size and ecosystem service provision may reflect the greater ecological complexity and visibility of larger wetlands. Larger systems can support more diverse habitats and ecological processes, which can enhance regulating and supporting services (Hambäck et al., 2023). In contrast, the negative association between surrounding farmland and ecosystem service provision likely reflects the pressures that intensive agricultural practices place on aquatic ecosystems, including nutrient inputs and hydrological alteration (Potter et al., 2004). Reducing these pressures, for example through improved land management or buffering of wetland features, may therefore enhance the range of ecosystem services delivered by RSuDS within agricultural landscapes (Eric et al., 2022; Everard et al., 2022).

3.4.5 Rapid Assessment of Wetland Ecosystem Services (RAWES) evaluation

The combination of expert knowledge and a simple assessment process make the RAWES framework a quick and low-cost method for evaluation of wetland ecosystem services. It is particularly useful for RSuDS because of the current lack of data associated with the ecosystem services they provide (Reed et al., 2020). This makes it accessible to a wide range of stakeholders, including local communities, non-governmental organizations, and government agencies. RSuDS, and wetlands more broadly, have historically been overlooked and undervalued, so having a readily accessible approach for assessing ecosystem services is crucial to ensure their contributions are recognised. It is important to note, however, that RAWES is intended as an initial scoping tool rather than a method for detailed quantification. Instead of producing precise measurements, RAWES uses indicative impact scores to capture the relative significance of ecosystem services, integrating multiple value systems and stakeholder perspectives. This simplified approach allows assessments to be applied across a larger number of sites and wider spatial areas, prioritising coverage and comparability over intensive measurement. As such, RAWES serves as a practical entry point for identifying and highlighting key services and trade-offs, while reserving more detailed, resource-intensive assessments for locations or services where greater precision is warranted.

A clearer understanding of the full range of ecosystem services delivered by NbS, including those that are difficult to quantify, is essential for informing land management and environmental policy. Standardised assessment approaches that incorporate both quantitative and qualitative dimensions can improve the consistency and comparability of ecosystem service evaluations, supporting more effective planning and implementation of NbS interventions (McVittie et al., 2023).

3. 5 Conclusion

This study demonstrates that RSuDS can generate multiple ecosystem services within agricultural landscapes, particularly regulating services associated with water quality improvement and flow regulation. Cultural services were more difficult to capture, while provisioning services such as food and water were less frequently observed. These findings highlight the multifunctional role that ponds and wetlands can play in supporting ecosystem functioning and societal benefits.

The results also demonstrate that the spatial context of RSuDS influences ecosystem service outcomes. Larger systems, those located closer to beneficiaries, and those surrounded by less intensive agricultural land were associated with higher ecosystem service provision. Strategic placement and design of RSuDS within agricultural landscapes can therefore enhance the benefits they deliver while helping to minimise potential trade-offs. These insights provide evidence-based guidance that may support the integration of ponds and wetlands into agri-environment schemes, catchment management strategies, and wider landscape planning.

The study also identifies several priorities for future research. In particular, uncertainty remains regarding the net climate regulation role of RSuDS, as small water bodies may both sequester carbon and emit greenhouse gases. Addressing this knowledge gap will require integrated assessments that measure greenhouse gas emissions and carbon burial simultaneously across different environmental conditions. Further work is also needed to improve methods for assessing cultural ecosystem services and to better understand how ecosystem service trade-offs vary across spatial and temporal scales.

By strengthening the evidence base on the ecosystem services delivered by ponds and wetlands, this research contributes to a better understanding of how nature-based solutions can support more sustainable and resilient agricultural landscapes.

4. Methane and carbon dioxide emissions from agricultural ponds are driven by physico-chemical and morphological characteristics

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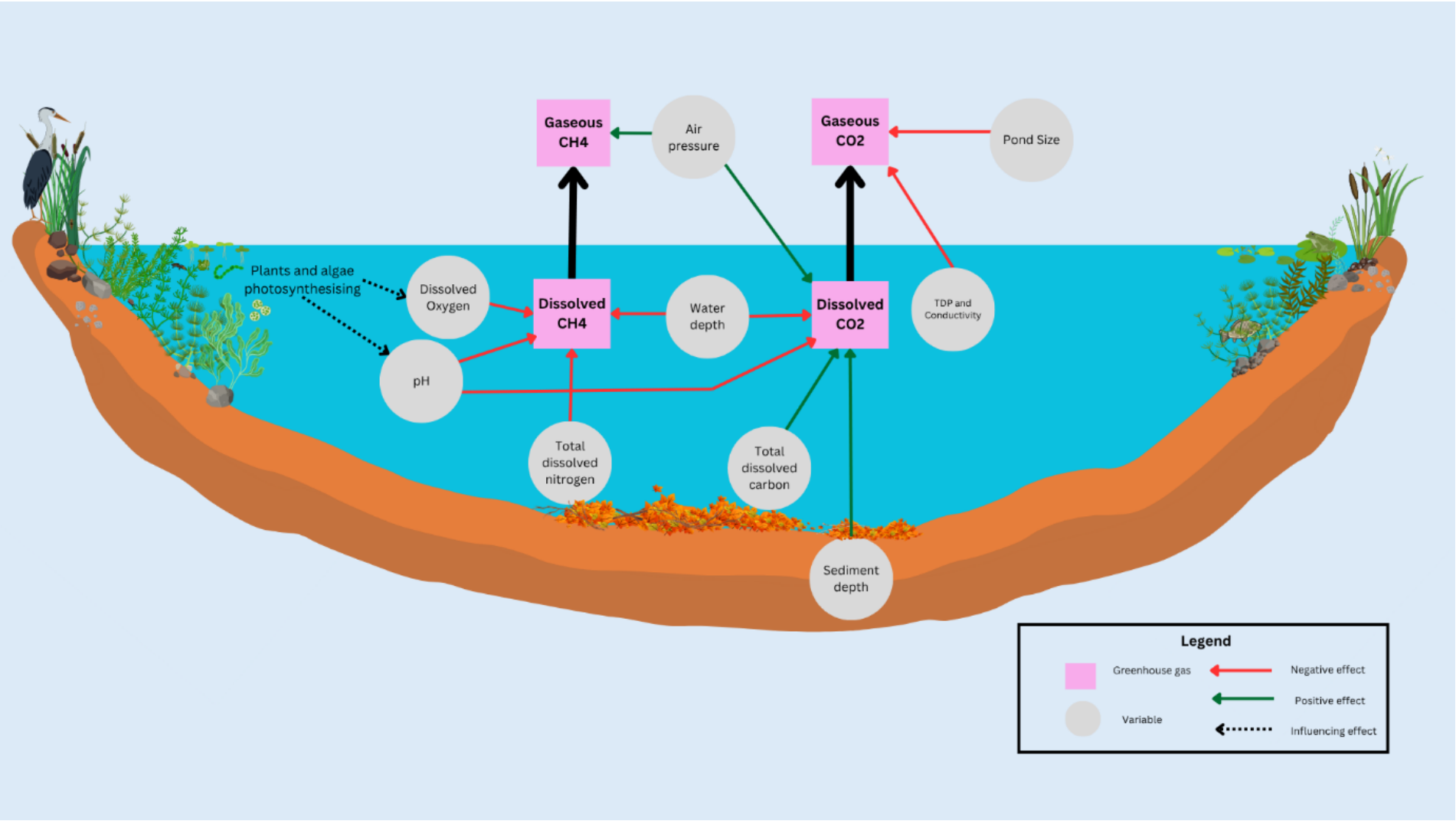
Narrative author contribution statement

I led the design and development of this research, including formulating the research questions and methodological approach to quantify greenhouse gas emissions from agricultural ponds. I designed and conducted the field sampling campaign across 18 ponds, including the collection of dissolved gas and associated physico-chemical data. I was responsible for data curation and carried out the formal analysis, including the calculation of gas fluxes and exploration of relationships with environmental drivers. I developed the analytical workflows, software tools, and visualisations used in this chapter. I led the interpretation of the results, particularly in relation to climate impacts and policy relevance, and wrote the manuscript, incorporating feedback from co-authors throughout the revision process.

Abstract

Agricultural ponds are globally widespread and multifunctional, yet they emit substantial quantities of greenhouse gases (GHGs), notably methane (CH₄) and carbon dioxide (CO₂), posing a significant but poorly quantified climate disservice. This study quantified dissolved concentrations and total fluxes of CH₄ and CO₂ from 18 agricultural ponds in Scotland between May and September. All ponds were net GHG sources, with mean fluxes of $0.39 \pm 0.41 \text{ g CH}_4 \text{ m}^{-2} \text{ day}^{-1}$ and $3.22 \pm 1.91 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$. While CO₂ flux dominated in absolute terms ($11.4 \pm 5.9 \text{ t/ha/year}$) CH₄ had a far greater climate impact when expressed as sustained global warming potential ($36.5 \pm 35.5 \text{ t CO}_2\text{-eq/ha/year}$). CH₄ fluxes peaked in warmer months, while CO₂ patterns were more variable. Dissolved CH₄ decreased with depth, pH, oxygen, and nitrogen, while dissolved CO₂ decreased with depth and pH but increased with air pressure, carbon, and sediment depth. As human-made ponds are classed as anthropogenic systems, their emissions should be included in national GHG inventories and land-based mitigation strategies. These findings highlight the importance of integrating agricultural pond emissions into climate policy and carbon budgeting.

Graphical abstract



4.1 Introduction

More than 95% of the world's waterbodies are small (under 10 hectares) with an estimated 547 million to 3 billion ponds spread across the globe (Downing, 2010; Verpoorter et al., 2014; Messenger et al., 2016). While many ponds form through natural geological processes, their value for irrigation, livestock watering, nutrient retention, and habitat provision has led to widespread construction, modification, and restoration of ponds by humans, particularly within agricultural landscapes (Schwartz and Miller, 2019). For example, in the United States there are an estimated 2.6 million ponds in agricultural areas, with a total pond area of approximately 420,000 hectares (Smith et al., 2002). Ponds are increasingly being recognised as important Nature-based Solutions (NbS) in creating multifunctional landscapes that are resilient to climate change (Cuenca-Cambronero et al., 2023). Agricultural ponds do this by providing a diverse range of ecosystem services which have ecological and societal value (Fu et al., 2018). The societal benefits of agricultural ponds can be divided into four categories: 1. Provisioning services, such as water supply for irrigation and livestock (López-Felices et al., 2020); 2. Regulating services that include flood mitigation, sediment trapping and water quality improvement (Robotham et al., 2021); 3. Cultural services through enhancing rural landscapes and supporting recreational activities (Fu et al., 2018); and 4. Supporting services, such as maintaining biodiversity and providing habitats for plants, invertebrates, amphibians, and waterbirds (Hill et al., 2021).

However, agricultural ponds also generate a significant ecosystem disservice as sources of greenhouse gases (GHG). Inland surface waters are major contributors to global emissions of carbon dioxide (CO₂) and methane (CH₄) (Cole et al., 2007; Raymond et al., 2013; Tranvik et al., 2009; Rosentreter et al., 2021). Small lentic systems, which includes ponds, release large quantities of GHG relative to their size due to rapid biogeochemical cycling, shallow depth, high productivity and strong benthic-pelagic coupling (Richardson et al., 2022). Ponds are estimated to account for 15% and 41% of CO₂ and CH₄ emissions, respectively, from inland surface waters despite comprising only 9% of lentic surface waters globally (Holgerson & Raymond, 2016). CH₄ is emitted from ponds via two distinct pathways: diffusion, where dissolved CH₄ passes across the air–water interface, and ebullition, where bubbles released from anoxic sediments transport CH₄ directly to the atmosphere. Most CH₄ studies in ponds focus on diffusive fluxes, yet ebullition can represent a substantial proportion of total CH₄ emissions (DeSontro et al., 2016; Holgerson and Raymond, 2016). In contrast, CO₂ is almost exclusively emitted via diffusion, as it remains dissolved in water and does not form gas bubbles under typical pond conditions. However, total CO₂ fluxes can still be highly dynamic and influenced by both biological processes (e.g. respiration and photosynthesis) and

physical factors (e.g. wind and temperature), with considerable temporal variability reported in small waterbodies (Webb et al., 2019; Holgerson et al., 2022).

Agricultural ponds generally exhibit higher CH₄ emissions than ponds in less disturbed or unmanaged environments, although CO₂ fluxes show greater variability depending on soil type and other environmental conditions (Peacock et al., 2021; Webb et al., 2023). This is largely due to elevated nutrient and organic matter inputs from runoff, fertilizers, and animal waste (Grinham et al., 2018; Martinez-Cruz et al., 2017). For example, urban-agricultural ponds in Belgium had CO₂ and CH₄ emission rates nearly three times higher than those of natural ponds (Bauduin et al., 2024). Between 2003 and 2019, global cropland area expanded by ~9%, much of it replacing natural ecosystems (Potapov et al., 2022). If ponds are present within anthropomorphic land uses, they will be exposed to pressures such as nutrient loading, sediment disturbance, and hydrological modification which can elevate GHG emissions, particularly CH₄, by stimulating microbial activity (Peacock et al., 2021; Holgerson & Raymond, 2016). In parallel, deterioration of water quality and oxygen levels can undermine the biodiversity and ecosystem services these ponds provide (Hill et al., 2021). GHG emissions from constructed agricultural ponds are classed as anthropogenic and can therefore be included within national greenhouse gas inventories and considered in land-based mitigation strategies. The 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories provides emission factors for both freshwater and saline constructed ponds, enabling their inclusion under the category Emissions from Flooded Land Remaining Flooded Land – Other Constructed Waterbodies (IPCC, 2019). Several nations have already incorporated ponds into their national inventories using these guidelines, including the United States (U.S. EPA, 2023) and Australia (DCCEEW, 2023). However, ponds remain underrepresented in global syntheses due to limited spatial and temporal data coverage and uncertainties in pond mapping (Lauerwald et al., 2023a).

GHGs in ponds are heterogeneous both spatially and temporally, therefore generalisation and upscaling are difficult (Webb et al., 2019; Bauduin et al., 2024). While several factors, such as nutrient levels, dissolved oxygen, temperature, vegetation cover, and pond morphometry can influence CO₂ and CH₄ emissions, their relative importance remains poorly known and context dependent (Ray et al., 2023; Vizza et al., 2022; Malyan et al., 2022). Agricultural ponds can undergo nutrient enrichment and frequent stratification, which can enhance CH₄ production and lead to large spatial and temporal variability (Ray & Holgerson, 2023; Webb et al., 2019). . As global data on pond GHG remain sparse (Malerba et al., 2022), identifying the key physico-chemical and morphological

drivers of GHGs is essential for improving emission estimates and reducing uncertainty in regional and global carbon budgets (Ray et al., 2023). To address knowledge gaps in the GHG dynamics of agricultural ponds, this study quantified CO₂ and CH₄ from 18 agricultural ponds across central Scotland. We aimed to: (1) quantify the concentrations and fluxes of CO₂ and CH₄ from agricultural ponds during the summer (May to September); (2) assess between month variability in GHG emissions from the ponds; and (3) identify the physico-chemical and physical characteristics regulating the GHG concentrations and fluxes from the ponds.

4.2 Methods

4.2.1 Agricultural Pond selection

18 agricultural ponds were studied. All waterbodies met the formal definition of ponds, < 5 hectares in area and < 5 m deep (Richardson et al., 2022), although most of the ponds in this study were substantially smaller than these thresholds. All ponds had < 30% emergent vegetation (with a mixture of emergent, submerged, and floating macrophytes) and permanent hydroperiods. All ponds were located within central Scotland and are low-lying (< 200 m). The predominant land use around these ponds was agricultural with a mixture of arable and grassland, both improved and unimproved (Figure 12). Most ponds (13) had inflows and outflows and were human-constructed, while five ponds lacked clear in- or outflows. Of these, four are likely of glacial origin, as they appear on the earliest available maps, and one is a human-developed spring-fed pond. Most ponds (15) were hydrologically independent, though three constructed ponds were hydrologically connected. While the precise origins of all ponds could not be confirmed, it is evident that most have been influenced or modified by agricultural practices over time. All ponds lie within the River Forth basin, which encompasses a range of soil types. In the lowland areas soils are predominantly noncalcareous gleys derived from estuarine and lacustrine raised beach silts and clays. In the higher areas, soils transition to brown earths formed from drifts derived mainly from sandstones with some basaltic components.

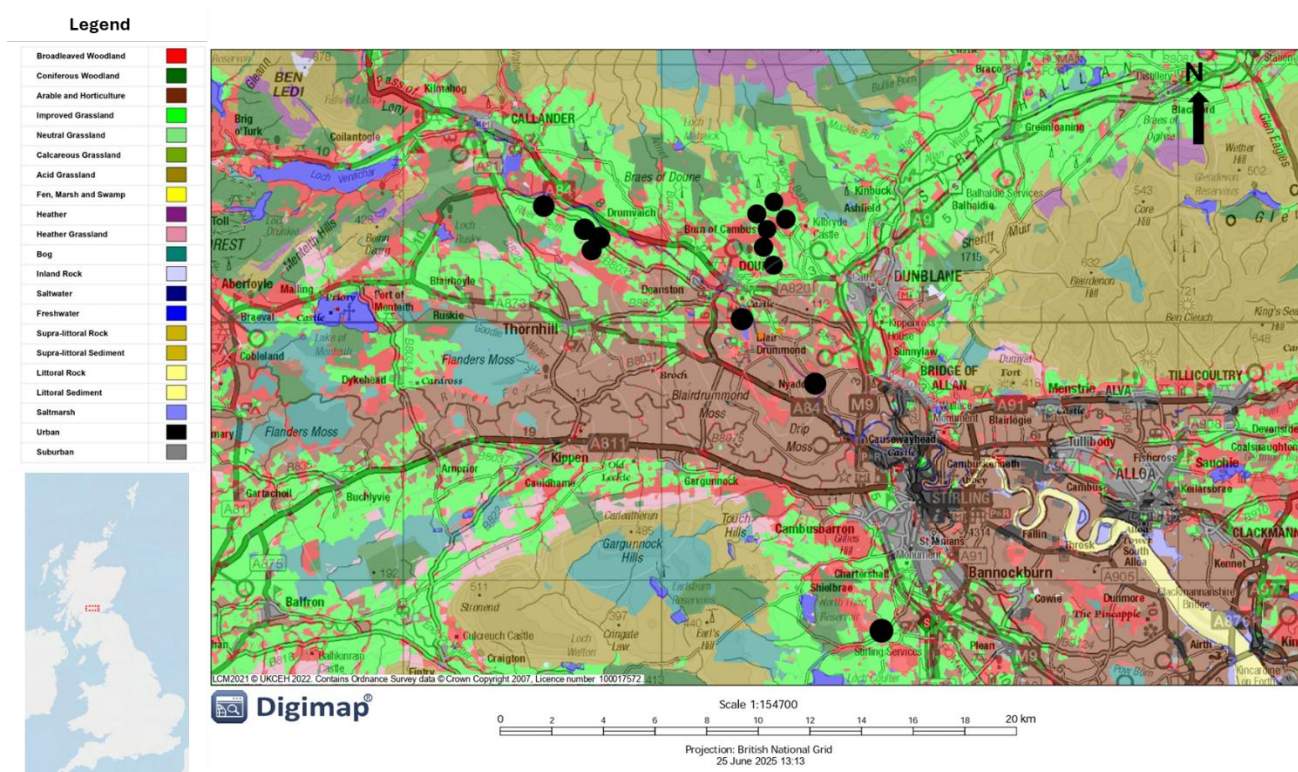


Figure 12: Location of ponds (black circles) with surrounding land use (CEH 2021).

4.2.2 Water sampling

Eight sampling events took place, once a month from May to September 2024, corresponding to the northern hemisphere summer. The ponds were sampled in a different order each month to avoid systematic bias. On each visit, water samples were taken from a different location around the perimeter of the pond to capture variation in space and depth (Ray et al., 2023). Two 500 mL water samples were collected: one from the edge of the pond and another approximately 2 m from the edge, using an extendable sampler. These samples were considered subsamples of the same pond at a single time point. To assess whether there were consistent spatial differences between edge and mid-pond samples, we compared the two positions for each water-quality variable across all ponds and sampling dates. Normality was evaluated separately for each variable using Shapiro–Wilk tests, all of which indicated non-normal distributions (all $p < 0.05$). Accordingly, Mann–Whitney U tests were conducted individually for each parameter. No statistically significant differences were detected between edge and mid-pond samples for any variable (all $p > 0.12$, $n = 301$), so the mean of the two positions was used and treated as a single sample for subsequent analyses. Water samples were collected from the ponds and immediately filtered in the field using a $0.45 \mu\text{m}$ cellulose syringe filter. The filtered samples were then transported in a cool box for storage and subsequent analysis. Most of the samples were analysed immediately for Total Dissolved Carbon (TDC), Dissolved Organic Carbon (DOC), Total Dissolved Nitrogen (TDN), and Total Dissolved Phosphorus (TDP) using a Total

Organic Carbon analyser (Elementar EnviroTOC, Germany). Dissolved Inorganic Carbon (DIC) was inferred by subtracting DOC from TDC. Of the 180 water samples collected, 39 (22%) were frozen immediately after collection and analysed within one month. Most of these samples ($n = 35$) were from the June sampling campaign due to laboratory equipment malfunction, with the remaining four from August. Freezing has been shown to have minimal effects on measured nitrogen and phosphorus concentrations, but reported effects on dissolved carbon vary among studies and systems. In temperate freshwater streams and wetlands, several studies have found that DOC concentrations are largely stable following freezing, with negligible or inconsistent changes depending on DOC composition and water chemistry (Peacock et al., 2015; Spencer et al., 2007; Heinz and Zak, 2018). Where effects have been observed, they are often small and variable in direction, making correction difficult to justify without system-specific validation (Fellman et al., 2008; Pokrovsky et al., 2018). Given this variability and the absence of clear evidence for systematic DOC loss in agricultural pond systems, no adjustment was applied to dissolved carbon concentrations in frozen samples. All results are therefore based on measured values. In addition to the water samples, a HACH HQ30d meter with probes for Dissolved Oxygen (DO), pH, and conductivity ($\mu\text{S}/\text{cm}$) was measured just beneath the water surface at the same locations where water samples were collected per visit.

4.2.3 Gas sampling

Sampling of GHGs took place on each visit at the same location as water sampling. Two complementary methods were used to quantify GHG dynamics in the ponds. Dissolved CO_2 and CH_4 concentrations were measured using the headspace equilibration technique (McAuliffe, 1971; Ray et al., 2023), capturing gases present in the surface water but not yet emitted. Floating chambers were used to collect total GHG flux samples, encompassing both diffusive and ebullitive components (Mannich et al., 2019). Using both methods enabled us to compare dissolved gas concentrations with actual emissions and to identify which environmental variables influenced different parts of the emission pathway, from in-pond production and storage to atmospheric release. Two samples were taken adjacent to each other at <15 cm depth from the pond edge. For each sample, 15 mL of ambient air was first drawn into a 60 mL syringe, followed by 45 mL of pond water from 5 cm below the water surface. The syringe was sealed, shaken vigorously for 1 minute to allow equilibration, and the 15 mL headspace gas was immediately transferred into a pre-evacuated 12 mL glass exetainer vial. Floating chambers were used to collect total flux samples (Mannich et al., 2019). Each chamber had a volume of 9500 cm^3 and a surface area of 1200 cm^2 . Three floating chambers were deployed over a 24-hour period and tethered to the shore with rope. This setup prevented chambers from drifting across the pond while still permitting some natural surface motion (Lorke et al., 2015;

Bastviken et al., 2004). A single gas sample (12 mL) was then taken from the headspace of each chamber. No explicit measurements of ebullition (i.e. bubble traps) were used but the floating chambers may capture bubbles in addition to diffuse flux. An ambient air sample was also collected per pond per visit, approximately 0.5 meters above the water surface at the pond edge. Syringes were flushed with air three times before collecting a sample. All samples were stored in pre-evacuated 12 mL glass exetainers. Gas samples were analysed at the University of Stirling by gas chromatography (Hewlett Packard GC 5890 Series II) using a flame ionization detector for measuring gas concentrations. The concentrations of CO₂ and CH₄ in the samples were quantified by comparing sample peak area against the standard curve of the peak areas of three different concentrations of known standards. The two samples from the headspace method and the three samples taken using the floating chambers were considered technical replicates and the mean CO₂ and CH₄ concentrations for each method were used in statistical analyses (Ray et al., 2023). Gas concentrations from the headspace samples were transformed into a dissolved concentration (Equation 2) using Henry's law and the ideal gas law (Ray et al., 2023). The samples from the floating chambers were transformed into a total daily flux (Equation 3) (Livingston and Hutchinson, 1995; Goeckner et al., 2022). The change in concentration (ΔC) was calculated from the difference between the ambient air sample and the averaged floating chamber concentration. Air temperature (T) and pressure (P) were acquired from a 24-hour average of nearby weather stations at similar elevations to the sampled ponds.

Equation 2: Formula used to calculate the concentration of dissolved gases in water (C_{water})

$$C_{water} = (P \times V_{air} \times \Delta C_{gas}) / (R \times T \times V_{water}) \times 10^6$$

Equation 3: Formula used to calculate daily greenhouse gas flux (F)

$$F = (\Delta C_{gas} \times V_{chamber} \times P) / (R \times T \times A \times \Delta t) \times 10^6$$

Where; ΔC_{gas} is Change in gas concentration ($\text{mol} \cdot \text{mol}^{-1}$), C_{water} is Concentration of GHG in water ($\mu\text{mol} \cdot \text{L}^{-1}$), V_{air} is Volume of air in the headspace (m^3), V_{water} is Volume of water sampled (L), $V_{chamber}$ is Chamber volume (m^3), P is Atmospheric pressure (kPa), R is Universal gas constant ($0.00831451 \text{ kPa} \cdot \text{m}^3 \cdot \text{mol}^{-1} \cdot \text{K}^{-1}$), T is Temperature in Kelvin (K), A is Surface area of the chamber (m^2), Δt is Time interval in days (d), F is Gas flux ($\mu\text{mol} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$), $\times 10^6$ is a conversion factor from mol to μmol .

To convert both the dissolved concentration and total daily flux from μmol to grams, they were multiplied by the molecular weight of the corresponding GHG. To allow comparison with other studies and inclusion in broader syntheses, seasonal GHG emissions (May–September) were extrapolated to annual estimates. This was done by applying the lowest observed monthly mean for

each GHG to the remaining unmeasured months (October–April). While this approach likely overestimates total annual emissions, it provides a consistent basis for comparison and a conservative upper-bound estimate in the absence of year-round measurements. This method follows similar practices used in other studies with partial seasonal coverage (Jensen et al., 2022; Peacock et al., 2019). The total annual flux of CH₄ was then transformed into CO₂-equivalents using the sustained global warming potential (SGWP) by multiplying it by 45 (Neubauer & Megonigal, 2015; Peacock et al., 2021). For August, 8 ponds were removed from the dataset due to instrumental error from the gas chromatograph.

4.2.4 Pond characteristics

The ages of the ponds were determined by asking the landowner or local farmers. For 9 ponds the age was not known, and we used a combination of historical maps and satellite imagery to determine the age to the closest 5 years. Three of the ponds pre-dated the earliest available Ordnance Survey maps (c. 1860). For these ponds, a minimum age of 165 years was assigned, corresponding to the limit of historical cartographic data. While these ponds may be older, their precise age could not be determined. At centennial timescales, pond physical structure is likely to have stabilised, such that modest differences in assumed age beyond this threshold are unlikely to influence our analyses. The area of the pond (m²) was measured using satellite imagery from Google Earth (2024). Water and sediment depths were measured from a canoe using a 5 m survey rod across an approximate 2 × 2 m grid for each pond. The rod was lowered until it contacted the sediment surface to record water depth, then gently pushed into the sediment until refusal, to determine sediment depth. Depth readings were subsequently mapped using geographic information systems (QGIS, 2025). The transect points for both water depth and sediment depth were interpolated using a triangulated irregular network to create a raster layer with 2 × 2 m pixels for each pond. We then calculated the average water and sediment depths for each pond by averaging all the raster pixels within the pond perimeter.

4.2.5 Environmental variables related to greenhouse gas emissions

We used piecewise Structural Equation Modelling (pSEM) to understand the relationship among environmental variables and GHG emissions from ponds (Table 8).

Table 8: Physico-chemical variables used in piecewise Structural Equation Modelling.

Variable	Collection method	Units
Dependent variables		
Diffuse flux CH ₄	Floating chambers	g/m/day
Diffuse flux CO ₂	Floating chambers	g/m/day
Dissolved CH ₄	Headspace	mg/L
Dissolved CO ₂	Headspace	mg/L
Explanatory variables		
pH	HACH HQ30d probe	N/A
Conductivity	HACH HQ30d probe	μS/cm
Dissolved oxygen	HACH HQ30d probe	mg/L
24hr average air temperature	Weather stations	Celsius
24hr average air pressure	Weather stations	hPa
Area of the pond	Google Earth and QGIS	m ²
Average sediment depth	Transects and QGIS	m
Average water depth	Transects and QGIS	M
Age of the pond	Historical maps and farmers	Years
Total Dissolved Nitrogen (TDN)	Water samples	mg/L
Total Dissolved Phosphorus (TDP)	Water samples	mg/L
Total Dissolved Carbon (TDC)	Water samples	mg/L

SEM enables the analysis of individual components of the GHG pathway and the environmental variables that drive GHG production and emission, while also allowing the identification of indirect effects and potentially missing pathways within the system (Mwanake et al., 2023). A conceptual SEM was developed which was based upon this pathway of GHG production (dissolved concentrations) and GHG emissions (total flux) with environmental variables acting upon a specific part of this pathway (Figure 13). The SEM was constructed based on causal relationships. For the dissolved model, physico-chemical and water quality parameters were selected along with sediment and water depth that would influence the production of CO₂ and CH₄ (Battin et al., 2008; Stanley et al., 2016; Mwanake et al., 2023). Air temperature and air pressure were chosen for the total flux model because they are key drivers influencing greenhouse gas concentrations and fluxes (Peacock et al., 2021) and pond area was considered because it is inversely related to emissions (Holgerson and Raymond. 2016). 24-hour mean values of air temperature and air pressure were used to avoid diurnal patterns of change.

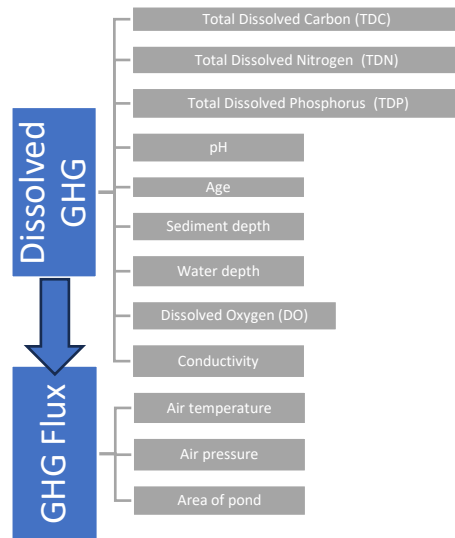


Figure 13: Conceptual structural equation model to illustrate direct and indirect relationships between environmental variables and dissolved GHG and GHG flux. Also shows how dissolved GHG is driving GHG flux.

4.2.6 Data exploration and analysis

Monthly variations between environmental variables were tested for significant differences using Kruskal–Wallis tests in R, implemented via the stats package (R Core Team, 2025). All pond characteristics (age, area, sediment depth, and water depth), were significantly positively correlated with one another (Figure 14). The strongest correlations were observed between pond age and mean sediment depth, and between mean water and sediment depth. Moderate correlations were found between pond age and area, and between area and mean water depth. The weakest correlations occurred between average water depth and pond age, and between pond area and average sediment depth. Given these correlations, we assessed multicollinearity prior to SEM. All variables were taken forward into the composite linear models used to construct the SEM, and

variance inflation factors (VIFs) were calculated for each. All VIF values were low (<4), indicating that multicollinearity was not a concern.

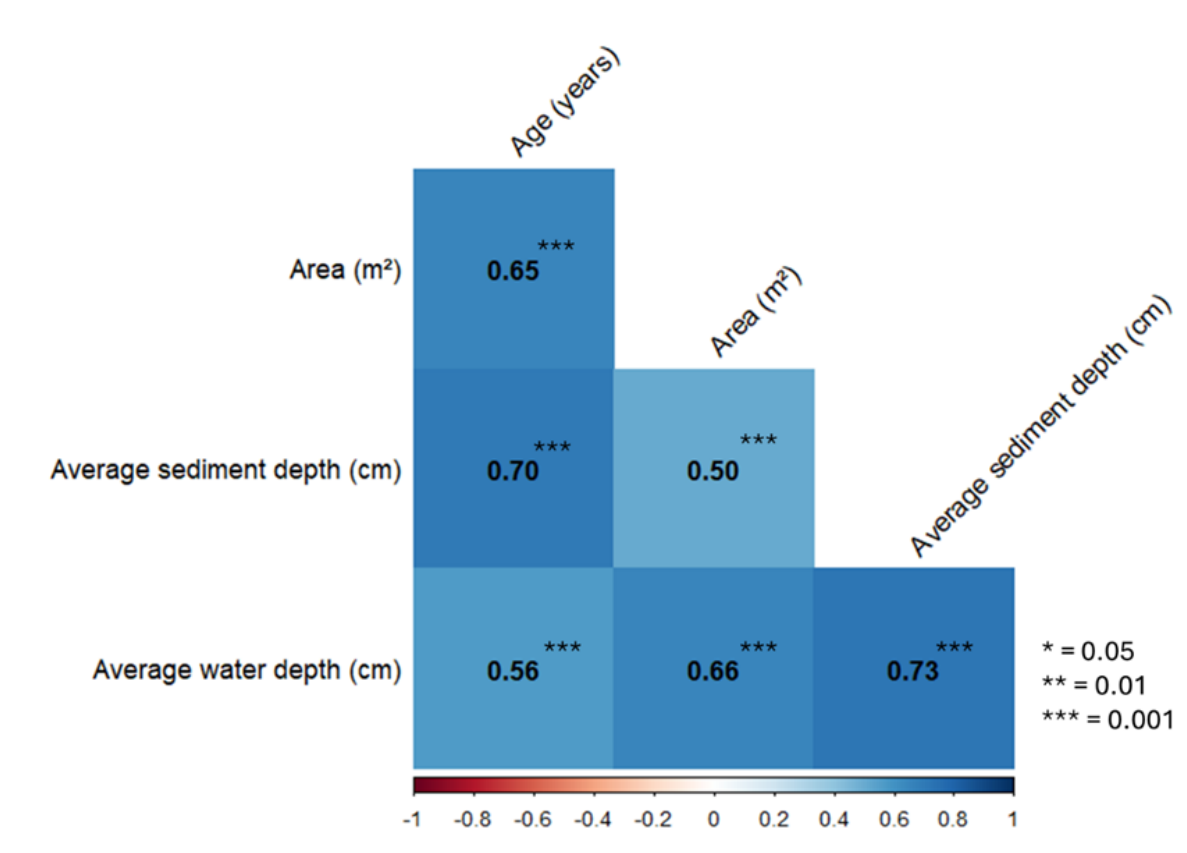


Figure 14: Correlations among pond characteristics of age, area, sediment depth and water depth, indicating correlation coefficients (R^2) and significance.

Prior to modelling, data were scaled and centred, and the dependent variables were transformed using Box–Cox transformations because the raw data were non-normally distributed; this improved normality and helped meet the assumptions of the linear models. Two linear models with Gaussian error distribution were then constructed, one for dissolved gases and one for total fluxes, using the lme4 package in R (Bates et al., 2015). Normality, heteroscedasticity, outliers, multicollinearity and temporal autocorrelation of the linear models were checked using the performance R package (Lüdtke et al., 2021). Spatial autocorrelation was checked with Moran’s I statistic (Bivand 2025) by using the X and Y coordinates of each pond. All models conformed to linear model assumptions, with no significant spatial autocorrelation found. We tested the inclusion of site as a random effect within linear mixed effect models but found that it masked the impact of unique site variables and enhanced the importance of sample-level variation. This is because the area of the pond, age, and average sediment and water depth were fixed each month whilst other environmental variables such as temperature and water quality fluctuated. The two LMs were then incorporated into a SEM,

which was run using the piecewiseSEM package (Lefcheck, 2016). Relationships not accounted for in our theorised SEM were checked using tests of directed separation within the same package. For the CO₂ SEM, conductivity and total dissolved phosphorus were found to have a greater influence on flux rather than dissolved concentrations, but no additional terms were added or removed. The best-fitting model was selected based on Fisher's C statistic and Akaike Information Criterion (AIC). All statistical analyses were undertaken using R (R Core Team, 2025) in RStudio (Posit Team, 2025).

3.3 Results

3.3.1 Pond characteristics

Pond physico-chemical variables showed considerable variation from May to September (Figure 15). Several variables exhibited statistically significant seasonal changes. Air temperature (Fig. 15.1) increased markedly across the summer, with significantly lower values in May and June and a peak in August, before declining in September. TDP (Fig. 15.5) showed a significant increase from May to August, followed by a slight decrease in September. TDC (Fig. 15.6) was significantly lower in May compared to all subsequent months, which did not differ significantly from one another. pH (Fig. 15.4) also varied significantly by month, with a notable decline in August, indicating that pond waters became less alkaline during peak summer.

In contrast, other variables showed no statistically significant monthly differences. DO, TDN, and conductivity (Fig. 15.3, 15.7, and 15.8) were relatively stable, with no significant changes across the sampling period, despite a decline in DO and TDN in August. Air pressure (Fig. 15.2) also showed no significant monthly variation, indicating consistent atmospheric conditions throughout the summer.

A full table including summary statistics of the pond characteristics and physiochemical variables can be found in Appendix F.

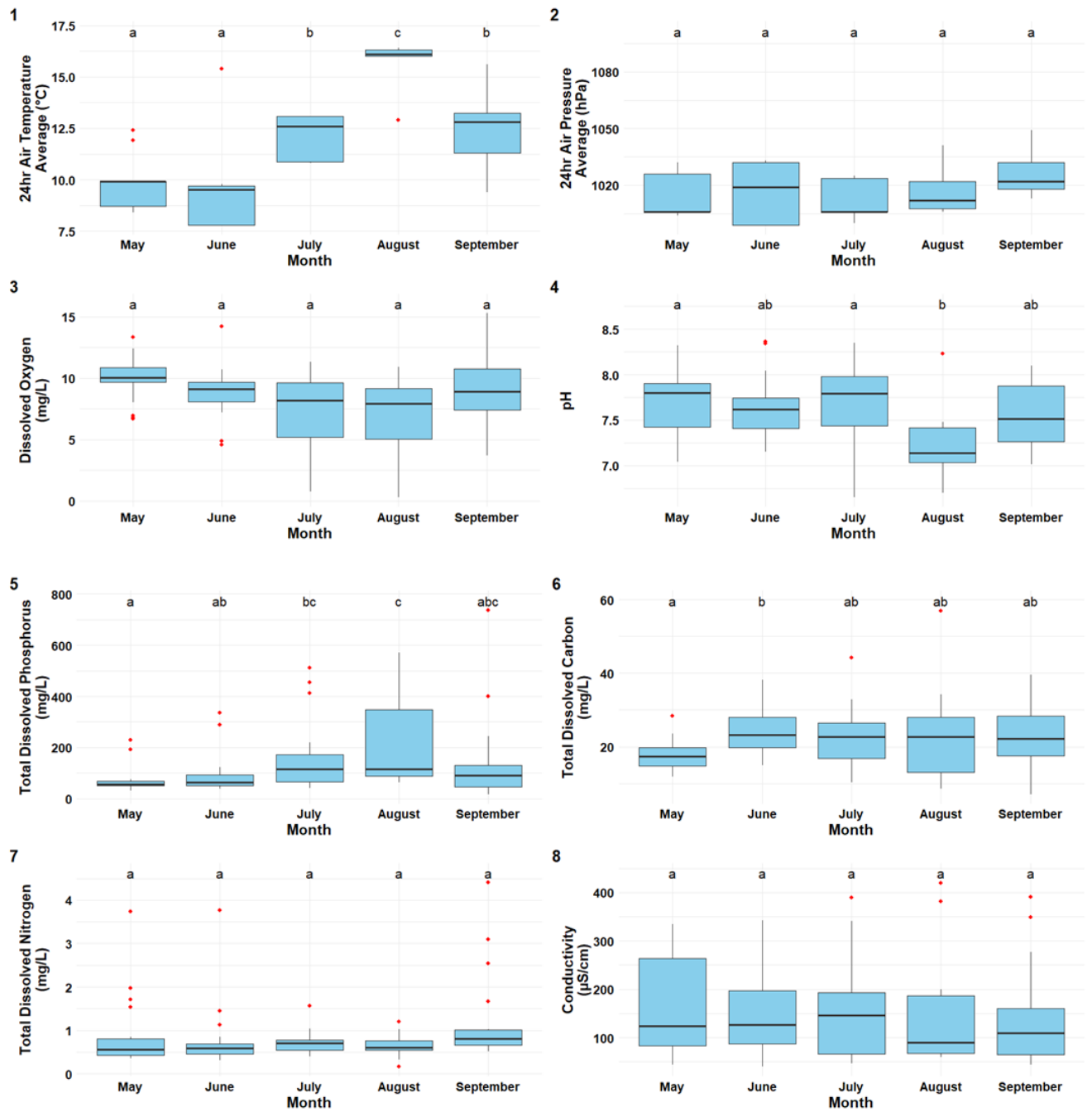


Figure 15: Seasonal variation in pond physico-chemical variables from May to September. Panels show monthly distributions for (1) 24-hour average air temperature (°C), (2) 24-hour average air pressure (hPa), (3) dissolved oxygen (mg/L), (4) pH, (5) total dissolved phosphorus (mg/L), (6) total dissolved carbon (mg/L), (7) total dissolved nitrogen (mg/L), and (8) conductivity (µS/cm). Letters above the boxes represent statistical groupings based on post-hoc comparisons: months that share the same letter are not significantly different from one another ($p > 0.05$), while months with different letters are significantly different ($p < 0.05$). For example, in panel 1, August is labelled “c” and is significantly warmer than months labelled “a” or “b”.

3.3.2 Seasonal variation of carbon dioxide and methane

Dissolved CH₄ concentrations and diffusive CH₄ fluxes both showed seasonal trends and variability, but not consistently (Figure 16). The mean dissolved CH₄ concentration (Fig. 16.1) across all months (May–September) was 0.0505 ± 0.0790 mg/L. Concentrations were relatively consistent from May through August, with higher variability during mid-summer, but declined in September. Boxplots indicate relatively narrow spreads in May and June, followed by greater variability in July and August, before concentrations decreased in September. However, statistical analysis showed no significant differences in dissolved CH₄ concentrations between months.

CH₄ fluxes (Fig. 16.2) had a seasonal mean of 0.387 ± 0.415 g/m²/day. Fluxes were relatively consistent across months, with a peak in July (0.471 ± 0.474 g/m²/day) and lower values in the surrounding months. In contrast to dissolved concentrations, the variation in flux was consistently higher across the season (Fig. 16.2), particularly notable in June and July. Despite these fluctuations in spread and median values, there were no statistically significant differences in CH₄ flux between months.

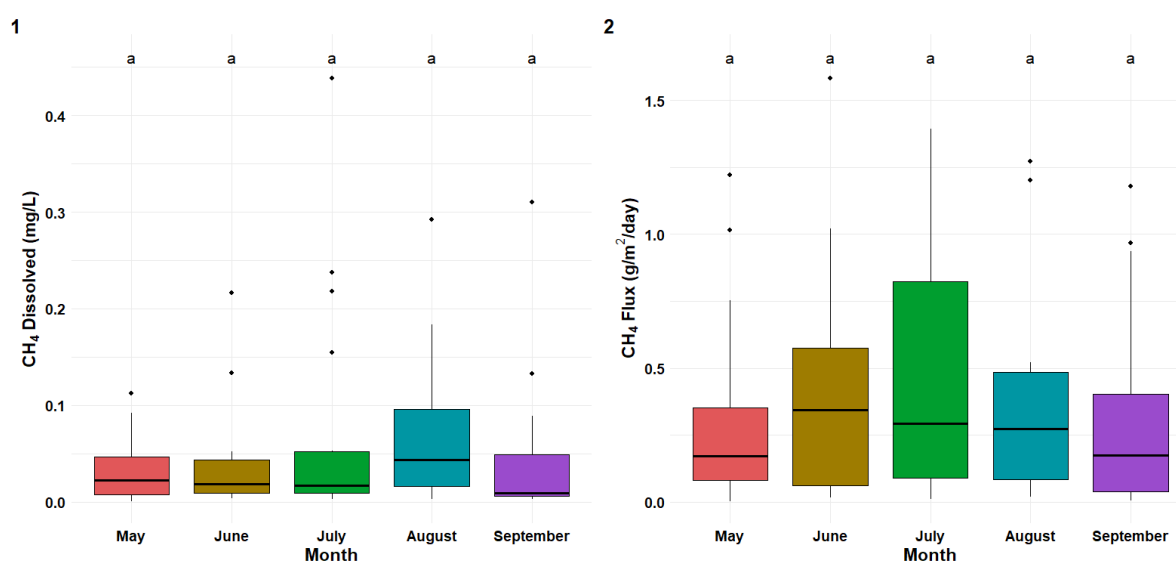


Figure 16: Dissolved CH₄ (1) and CH₄ flux (2) from May to September. Letters above the boxes represent statistical groupings based on post-hoc comparisons: months that share the same letter are not significantly different from one another ($p > 0.05$), while months with different letters are significantly different ($p < 0.05$).

Both CO₂ flux and dissolved concentration were higher and more variable than CH₄. There were no clear seasonal trends, but there was still large variation among months (Figure 17). The highest dissolved CO₂ concentrations were in September 1.15 ± 0.742 mg/L (Fig. 17.1), but no significant differences occurred among the months. The monthly pattern of diffusive CO₂ (Fig. 17.2) flux mirrored that of the dissolved concentrations but was more pronounced: September had the highest

flux (4.21 ± 1.83 g/m²/day), followed by June (3.55 ± 1.71 g/m²/day), with lower values in May, July, and August. However, despite the observed variability, statistical analysis indicated that neither dissolved CO₂ concentrations nor CO₂ flux differed significantly between months.

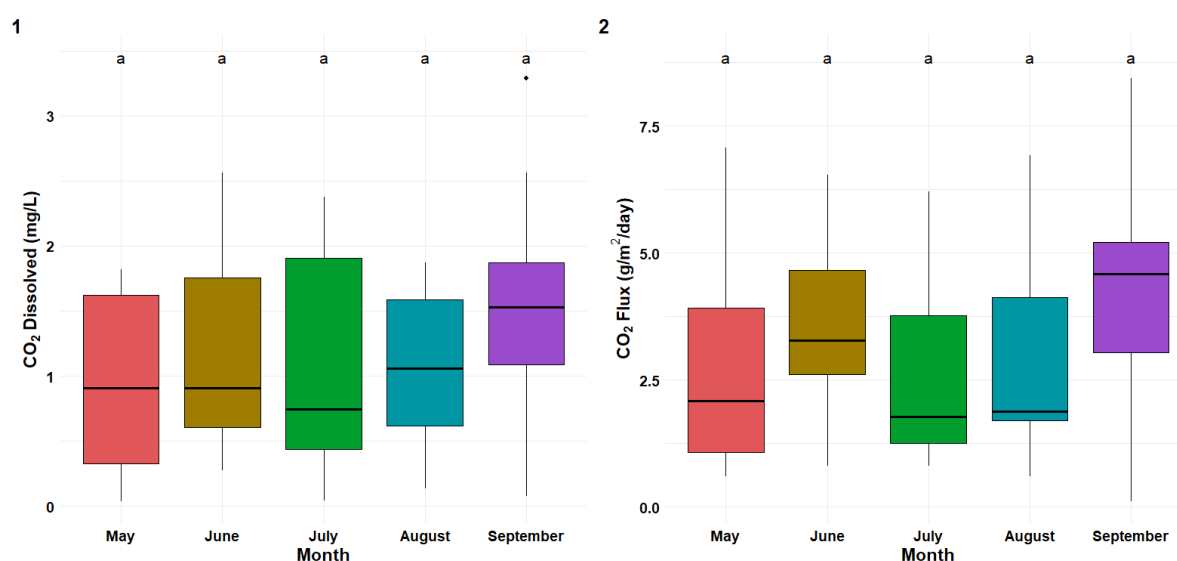


Figure 17: Dissolved CO₂ (1) and CO₂ flux (2) Measured once per month (May to September. Letters above the boxes represent statistical groupings based on Kruskal wallis test: months that share the same letter were not significantly different from one another ($p > 0.05$), while months with different letters are significantly different ($p < 0.05$).

The mean total annual flux for CH₄ was 0.811 ± 0.790 t/ha/year, whilst for CO₂ it was 11.4 ± 5.9 t/ha/year (Fig 18.1). However, when transformed into sustained global warming potential (CO₂-equivalents), CH₄ had a larger impact at 36.5 ± 35.5 t CO₂-eq/ha/year (Fig 18.2).

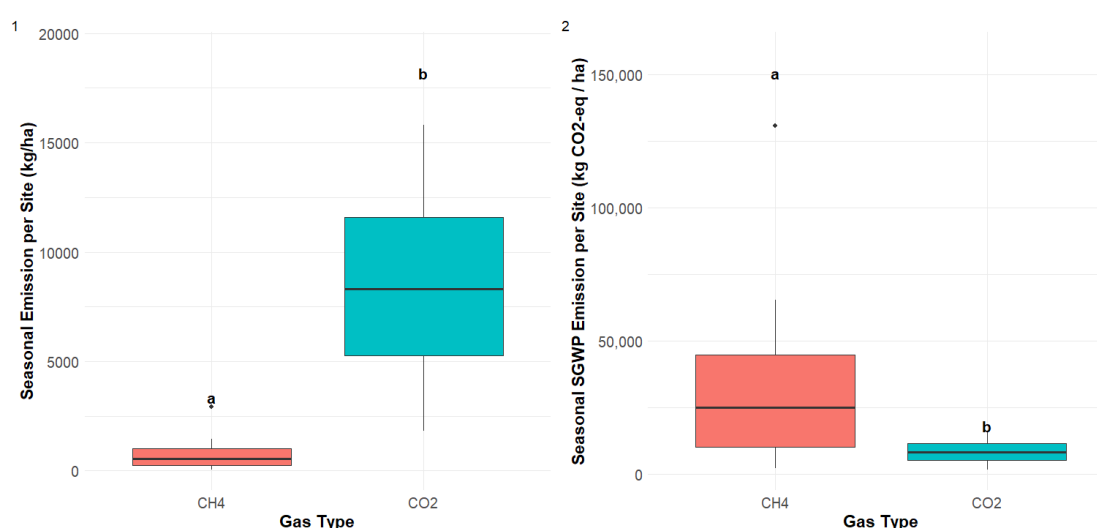


Figure 18: Total flux of CH₄ and CO₂ emitted from all 18 ponds from May – September 2024 (1). Panel 2 shows global warming potential per gas type. Letters above the boxes represent statistical groupings based on t-test: sharing the same letter indicates no significant difference ($p > 0.05$), while different letters are significantly different ($p < 0.05$).

Dissolved CH₄ concentrations were positively related to CH₄ total fluxes across all chambers and sampling occasions ($R^2 = 0.36$, $p < 0.001$; Fig. 19). This relationship is consistent with diffusive transport dominating methane emissions, as flux is proportional to the dissolved concentration gradient. However, substantial scatter around the regression line indicates that fluxes were not exclusively controlled by dissolved CH₄. Several observations exhibited disproportionately high CH₄ total fluxes relative to dissolved concentrations, suggesting episodic contributions from ebullition.

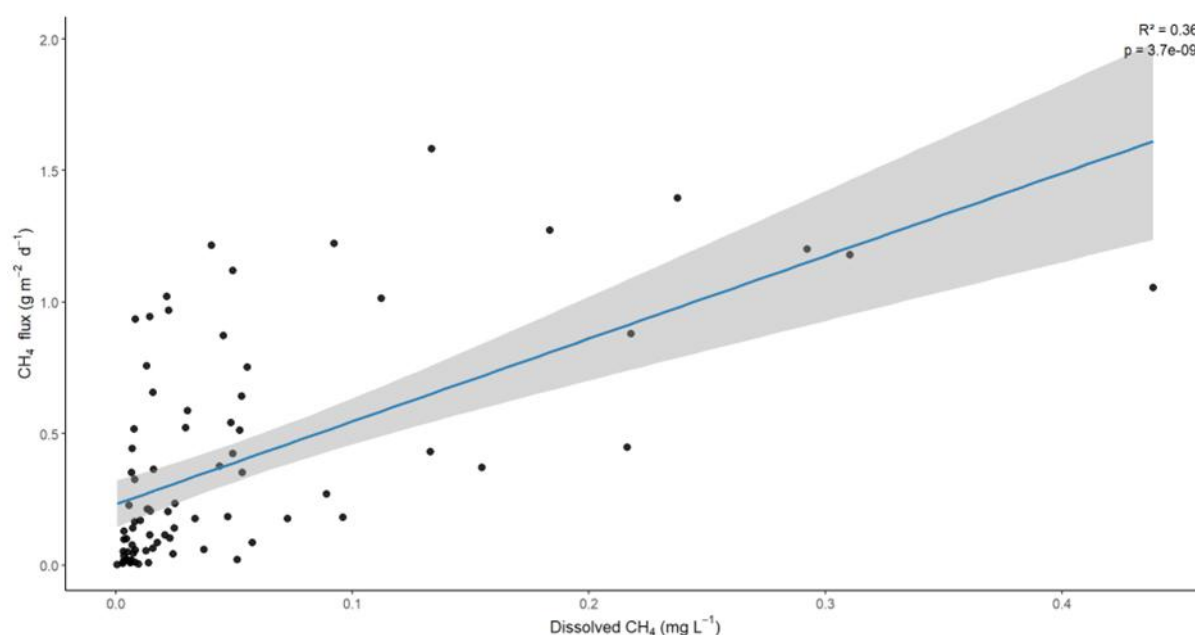


Figure 19: Scatter plot for CH₄ flux against dissolved CH₄ concentration for all the GHG samples taken. Trend line and R^2 value show the relationship between the two.

3.3.3 Structural equation modelling (SEM)

For both CH₄ and CO₂ (Figures 9 and 10), the goodness-of-fit statistics indicated that the models reproduced the data well and adequately captured the observed covariance structure (Fisher's $C = 21.4$, $df = 16$, $P = 0.163$ for CH₄; $C = 11.2$, $df = 12$, $P = 0.514$ for CO₂). The total explanatory power of the SEM was quantified using the coefficients of determination (R^2) of the endogenous variables. In the CH₄ model, the SEM explained 48% of the variance in dissolved CH₄ concentrations and 46% of the variance in gaseous total CH₄ fluxes. In the CO₂ model, the SEM explained 58% of the variance in dissolved CO₂ and 74% of the variance in gaseous total CO₂ fluxes. These values indicate moderate to high explanatory power, particularly for gaseous CO₂ emissions. While a full set of variables was included in the initial SEMs (Figure 2), only those with statistically significant paths are shown in the final diagrams for clarity. The full outputs, including both standardised and unstandardised effects for all variables, are provided in supplementary information (Appendix G). Dissolved CO₂ concentrations were strongly correlated with total CO₂ fluxes (Pearson's $r = 0.78$, $p < 0.001$), while

dissolved CH₄ concentrations showed a moderate but significant correlation with total CH₄ fluxes ($r = 0.60$, $p < 0.001$).

For CH₄ (Figure 19) it was observed that as pH, DO, TDN and water depth increased, there was a significant decrease in dissolved CH₄ in the water column. The strongest of these was DO followed by water depth, then TDN and finally pH. For diffuse CH₄ flux, there were two significantly positive drivers, with dissolved CH₄ having a primary effect and a minor effect from air pressure. For dissolved CH₄, 48% of the variance was explained whilst for diffuse CH₄ flux the model explained 46% of the variance.

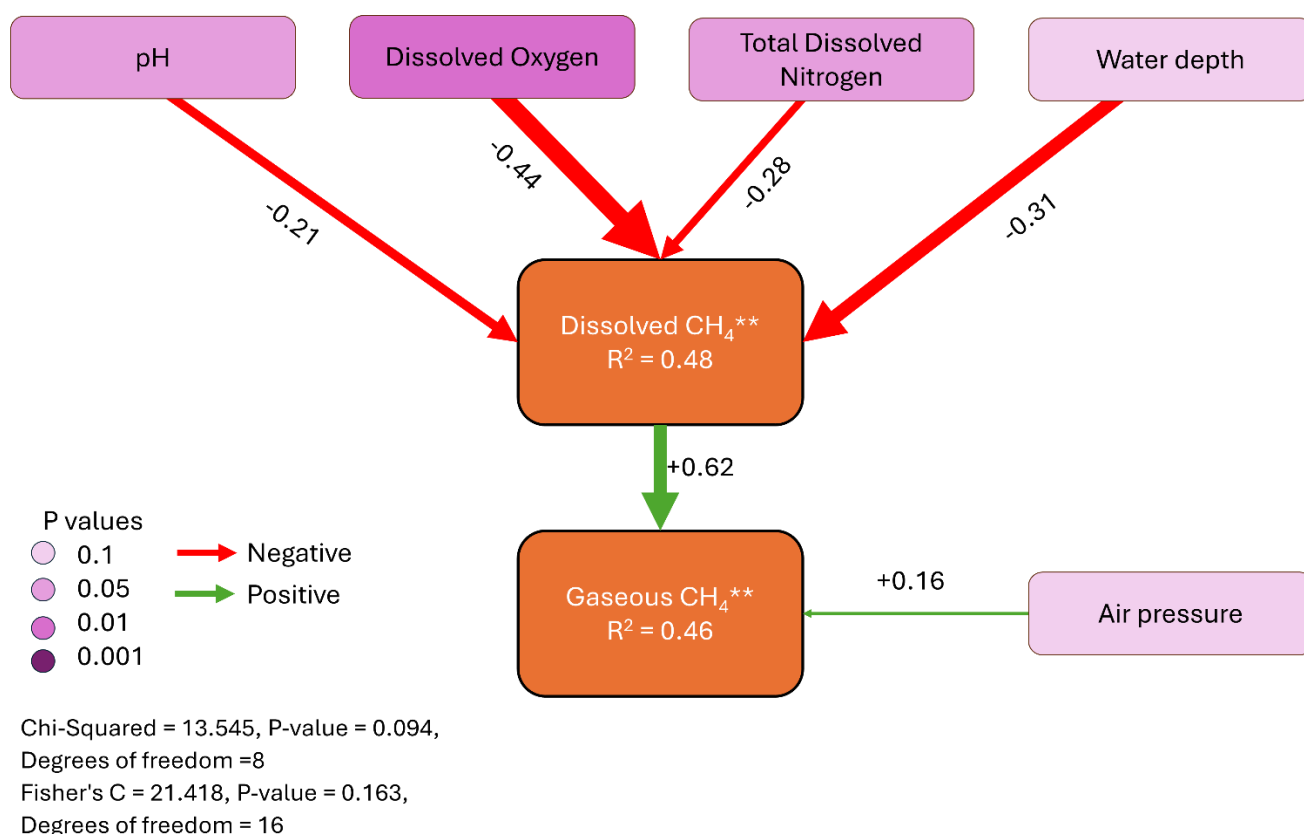


Figure 19: Structural equation model for CH₄. Significance values are identifiable by shading of pink/purple, with positive relationships shown in green and negative in red. The numbers attached to the arrows are the standardised estimates. Coefficients of determination (R²) are shown for each response variable.

Explanatory variables for each GHG were not consistent. For CO₂, a greater number of variables significantly influenced both the dissolved and flux components, with both positive and negative effects observed (Figure 20). Similar to CH₄, we found that as pH and water depth increased, dissolved CO₂ decreased, with water depth being the stronger predictor. In contrast to CH₄, we found strong positive relationships for sediment depth, air pressure, and a weakly positive

relationship for TDC. The primary driver of CO₂ flux was dissolved CO₂. Area of the pond had a strong negative effect followed by TDP and then conductivity. For dissolved CO₂, 58% of the variance was explained whilst for diffuse CO₂ flux the model explained 74% of the variance.

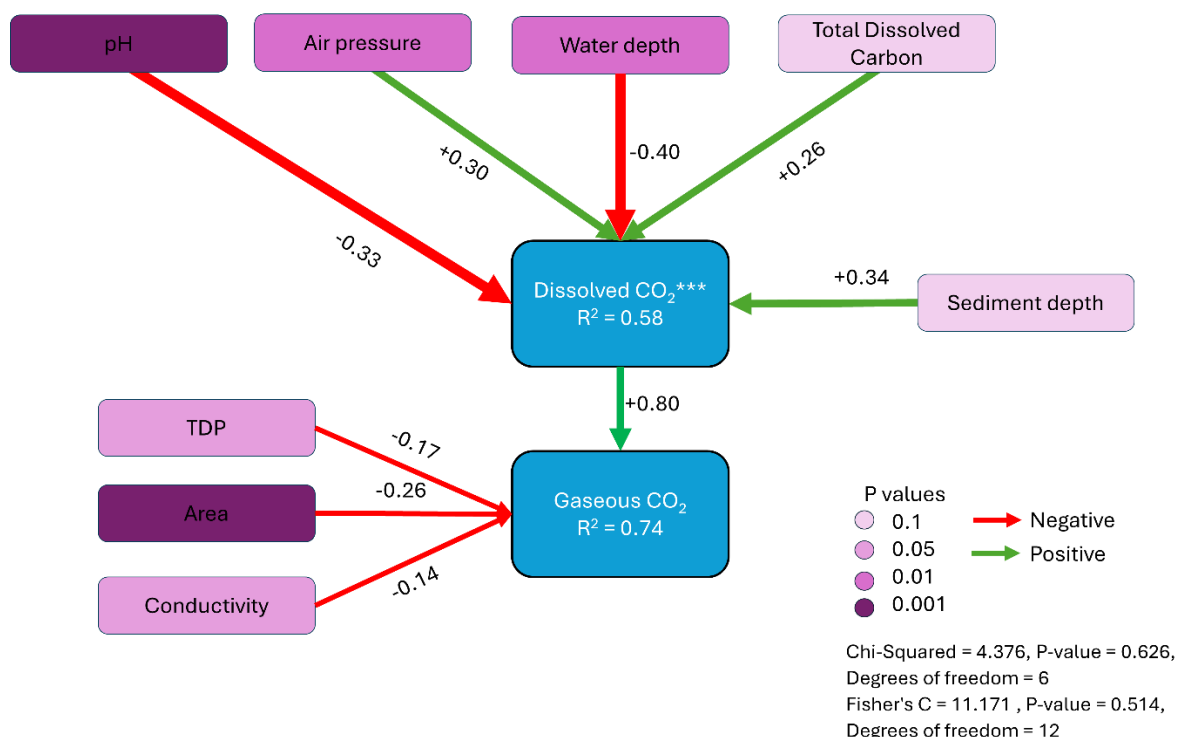


Figure 20: Outputs of the structural equation model for CO₂. Significance values are identifiable by shading of pink/purple. Positive relationships are shown in green and negative in red. The numbers attached to the arrows are the standardised estimates.

4.4 Discussion

Previous studies have demonstrated that emissions from small waterbodies are shaped by a range of factors, including surface area (Holgerson and Raymond, 2016), temperature, DOM supply, and trophic state (Peacock et al., 2021; Webb et al., 2019; Ollivier et al., 2019). Our results add to this evidence base by showing that CO₂ and CH₄ dynamics in agricultural ponds vary across the growing season and are influenced by a combination of physico-chemical conditions and pond morphology. By measuring both dissolved concentrations and emitted fluxes across 18 agricultural ponds from May to September, we provide new seasonal data that help refine understanding of GHG behaviour in these understudied systems and support their improved representation in regional and global carbon budgets.

4.4.1 Seasonal emissions from agricultural ponds

The majority of the GHG emissions from the agricultural ponds in this study was CO₂, whilst CH₄ was emitted in smaller quantities. However, when converted to CO₂-sustained equivalent global warming potential (SGWP), CH₄ was found to be higher. Similar findings have been reported from both managed and natural freshwater systems where CO₂ was the largest absolute flux, but CH₄ had the greater climatic impact (DelSontro et al., 2016; Holgerson and Raymond, 2016). Yet, other studies have found that CH₄ was the dominant flux (Ollivier et al., 2019; Peacock et al., 2021; Ray & Holgerson, 2023). Even ponds with low nutrient levels and minimal anthropogenic influence release substantial amounts of CH₄ (Holgerson and Raymond, 2016; Bastviken et al., 2011). This is due to inputs of nutrients and organic matter from the surrounding landscape, but agricultural ponds can experience intensified microbial activity that enhances GHG production (Beaulieu et al., 2019; Downing et al., 2008). Elevated nutrient and organic matter inputs promote eutrophication, leading to algal and bacterial blooms that increase microbial respiration and rapidly deplete DO. This creates anoxic conditions in bottom waters and sediments, which in turn favour methanogenesis and incomplete denitrification, thereby increasing methane emissions (Harrison et al., 2021; Malyan et al., 2022; Rabaey and Cotner., 2022).

The observed higher variability of CH₄ compared to CO₂ emission rates in this study, is likely due to the higher spatial heterogeneity of CH₄ which is due to the episodic and unpredictable nature of ebullition (Wik et al., 2016). Ebullitive CH₄ emissions are highly sensitive to localized sediment conditions, temperature, and pressure changes which can result in large differences between nearby ponds or within the same pond (DelSontro et al., 2016; Natchimuthu et al., 2017). Contrastingly, CO₂ emissions are predominately controlled by diffuse pathways, influenced by respiration, decomposition, and primary production, which vary less at small spatial scales (Holgerson and Raymond, 2016). The lower explained variability (R²) values of the CH₄ SEM compared to the CO₂ SEM possibly reflects this difference in emission pathways.

Throughout the summer (May–September), all the agricultural ponds in this study functioned as net GHG emitters. This seasonal pattern is consistent with observations from many nutrient-rich and agriculturally influenced freshwater systems, where inputs of nutrients and organic matter sustain elevated microbial respiration and methanogenesis during warm periods (DelSontro et al., 2016; Beaulieu et al., 2019; Natchimuthu et al., 2017). However, freshwater systems, including ponds, can show substantial seasonal variability, with periods of net CO₂ uptake when primary production exceeds respiration (Holgerson and Raymond, 2016; Sieczko et al., 2016). Indeed, several studies report that ponds may act as CO₂ sinks during phases of intense autotrophic activity, even if they

remain substantial CH₄ sources overall (Webb et al., 2019; Ray et al., 2023; Wik et al., 2014; Holgerson et al., 2022). In our study, peak total CH₄ fluxes occurred during the warmest months, aligning with previous work showing that CH₄ emissions are typically highest in summer when water temperatures are elevated (Ray and Holgerson, 2023b; Bauduin et al., 2024). Although temperature is often correlated with GHG emissions, it may not act as a direct driver; rather, its influence is mediated through temperature-sensitive processes such as microbial metabolism, macrophyte growth, stratification dynamics, and organic matter turnover (Zeikus and Winfrey, 1976; Wik et al., 2014; Yvon-Durocher et al., 2014; Davidson et al., 2018). While air temperature was not included in final SEMs, it likely plays an indirect role in shaping seasonal emission patterns through these biological and physical mechanisms.

CO₂ emissions from the agricultural ponds followed a more complex seasonal pattern than CH₄. Lower emissions were found during the months of July and August with the peak of emissions in September. This pattern reflects the seasonal dynamics of aquatic primary production and respiration and has been found in other small freshwaters ponds where the rapid growth of macrophytes and phytoplankton in the warmer months drives high rates of photosynthesis, resulting in high rates of uptake of dissolved CO₂ from the water column (Grasset et al., 2020; Ray and Holgerson, 2023b). By September, macrophytes in the agricultural ponds have started to senesce, driven by decreasing light availability and lower temperatures. This results in the accumulation and decomposition of autochthonous organic matter (Bartosiewicz et al., 2021). In addition, several ponds received inflows from the surrounding agricultural landscape, contributing allochthonous organic matter to the system. Together, these OM sources increase the amount of degradable carbon entering sediments toward the end of the growing season, promoting heterotrophic respiration and the observed rise in CO₂ emissions (Bauduin et al., 2024). The elevated fluxes observed in June are more difficult to explain, as they occurred during a time when active photosynthesis should still be occurring. One possibility is that this peak represents a lag in primary production relative to an early seasonal increase in respiration or an external input of organic material that temporarily stimulated microbial CO₂ production. Several ponds in the study have inflows from surrounding intensively managed arable and pasture land, where agricultural practices such as the application of synthetic fertilisers (e.g. urea or ammonium nitrate) or the spreading of manure and slurry can lead to increased nutrient and organic carbon loading in adjacent waterbodies. These inputs have been shown to stimulate microbial respiration and enhance CO₂ emissions in aquatic systems (Malyan et al., 2022; Goeckner et al., 2024). Future studies investigating temporal changes in GHG dynamics would benefit from concurrent, high-frequency measurements of water temperature, photosynthesis and respiration (e.g. continuous DO

monitoring), as these processes strongly influence carbon cycling in small waterbodies (Rabaey et al., 2024).

Our annual CH₄ and CO₂ flux estimates align with, and in many cases exceed, those reported in comparable studies of small temperate ponds. The mean annual CH₄ flux observed in our study (811 ± 790 kg CH₄ ha⁻¹ yr⁻¹) is substantially higher than the 2019 IPCC Refinement emission factor for constructed freshwater ponds (183 kg CH₄ ha⁻¹ yr⁻¹; IPCC, 2019), and also surpasses estimates from other regions, including Australia (~238 kg CH₄ ha⁻¹ yr⁻¹; Grinham et al., 2018), Europe (~50–300 kg CH₄ ha⁻¹ yr⁻¹; Peacock et al., 2021), and urban ponds in Brussels (~200–300 kg CH₄ ha⁻¹ yr⁻¹; Bauduin et al., 2024). Similarly, our CO₂ emissions ($11,400 \pm 5,900$ kg CO₂ ha⁻¹ yr⁻¹) are within the broad range reported across small pond studies (e.g. Holgerson & Raymond, 2016; Jensen et al., 2023; Webb et al., 2019). The dominance of CH₄ in the overall warming potential we observed is also consistent with previous findings that CH₄ contributes disproportionately to total GHG emissions in small lentic systems (Ollivier et al., 2019; Peacock et al., 2021; Ray et al., 2023). These comparisons reinforce growing recognition that small ponds are major contributors to inland water GHG budgets and should be more explicitly represented in syntheses and national inventories (Lauerwald et al., 2023).

Despite clear seasonal patterns in CH₄ emissions, air temperature did not emerge as a significant predictor of either CH₄ or CO₂ total fluxes in the SEMs. This contrasts with a substantial body of literature identifying temperature, particularly water and sediment temperature, as a key control on aquatic CH₄ production and emissions (Yvon-Durocher et al., 2014; Lauerwald et al., 2023a). Methanogenesis is known to be highly temperature sensitive, especially in sediment layers where microbial activity increases rapidly with warming (Yvon-Durocher et al., 2014).

The absence of a significant temperature signal in our models may reflect the relatively narrow thermal range captured during the summer sampling period, when temperatures are consistently high and therefore unlikely to be limiting for microbial activity. Under such conditions, other factors, including organic substrate availability, redox conditions, and pond morphology, can exert stronger control over GHG emissions than temperature alone (Bastviken et al., 2004; DelSontro et al., 2016; Goeckner et al., 2022). Temperature may therefore influence emissions indirectly, or at temporal scales not resolved by monthly summer sampling, particularly through interactions with mixing dynamics, oxygen availability, and carbon supply.

4.4.2 Drivers of emissions from agricultural ponds

While seasonal patterns revealed important temporal dynamics in CO₂ and CH₄ emissions from agricultural ponds, understanding the underlying environmental influences is essential for explaining the observed variability and for improving the representation of agricultural ponds in regional and GHG assessments (Ray et al., 2023).

4.2.1 Physical pond characteristics

The SEMs highlighted that increased water depth was associated with lower dissolved concentrations of both CO₂ and CH₄. This inverse relationship may reflect reduced vertical mixing in deeper ponds, which can limit the transfer of gases produced in bottom waters to the surface (Ray and Holgerson, 2023b; Søndergaard et al., 2023). Although we did not measure stratification directly, it is possible that thermal or chemical stratification intermittently restricted gas exchange, leading to lower surface concentrations and contributing to differences between dissolved and emitted GHG measurements.

In this study, pond area and water depth were positively correlated. Deeper ponds had greater water volumes into which gases could dissolve, potentially diluting surface concentrations (Holgerson, 2015). For CH₄ produced at depth, this dilution effect applies only to dissolved gas, and any deeper CH₄ would reach the floating chambers primarily via ebullition rather than diffusion. In addition, deeper waters are typically colder, which can reduce sediment methanogenesis, and provide a larger water column within which methanotrophy can occur, further suppressing CH₄ accumulation near the surface (Crevecoeur et al., 2017). Greater water depth can also promote more complete oxidation of organic matter before it reaches the sediments, limiting substrate availability for methanogenesis (Goeckner et al., 2022). The SEM also highlighted that CO₂ emissions declined as pond surface area increased, which is consistent with the broader literature (Lapierre and del Giorgio, 2012; Kankaala et al., 2013; Raymond et al., 2013; Holgerson and Raymond, 2016; Borges et al., 2022). Smaller waterbodies, such as ponds, typically have shallower water, higher sediment volumes, and larger perimeter-to-volume ratios, which promote frequent mixing, enhance benthic–pelagic coupling, and support the rapid accumulation of organic carbon (Goeckner et al., 2022; Wetzel, 2001). They are also more sensitive to surrounding land use, and their extensive perimeter-to-area ratios facilitate proportionally greater terrestrial carbon loading (Maberly et al., 2013; Holgerson and Raymond, 2016). These characteristics contribute to elevated CO₂ and CH₄ concentrations in shallow waterbodies (Deemer and Holgerson, 2021).

4.2.2 Physico-chemical parameters

We found that as the pH of pond water increased, the concentrations of both dissolved CO₂ and CH₄ decreased. For CO₂ this relationship is likely driven by biological processes, particularly photosynthesis by algae and macrophytes, which remove CO₂ from the water column and elevate pH (Goeckner et al., 2022). The uptake of CO₂ by autotrophs during photosynthesis shifts the carbonate equilibrium, reducing the proportion of free CO₂ and increasing the prevalence of bicarbonate and carbonate ions at higher pH (Stets et al., 2017; Malyan et al., 2022). Supporting this, we also found a negative relationship between conductivity and emitted CO₂, with conductivity likely serving as a proxy for underlying carbonate chemistry and ionic strength. Elevated conductivity may indicate the presence of bicarbonate-rich waters with greater buffering capacity, which stabilizes pH and reduces the partial pressure of free CO₂ available for emission (Stumm and Morgan., 2013).

During the sampling (May to September), both pH and DO exhibited a declining trend, with the lowest values observed in August. This is somewhat counterintuitive, as primary productivity is typically expected to peak in summer, increasing both DO and pH. The observed decline suggests a decoupling of photosynthesis and gas dynamics, possibly due to phytoplankton senescence, increased respiration, or oxygen consumption exceeding production during warm, stratified conditions. This aligns with previous findings where late-summer conditions led to oxygen depletion and acidification, especially when organic matter decomposition intensified (Bauduin et al., 2024; Bartosiewicz et al., 2021). The inverse relationship between pH and CH₄ may also reflect the influence of macrophytes and oxygen availability. Submerged and floating vegetation can lower CH₄ through root-mediated oxygen transport, which supports methanotrophs, and through microbial biofilms that filter CH₄ (Bastviken et al., 2023). In this study, as DO increased, CH₄ concentrations decreased, supporting prior observations of enhanced CH₄ oxidation under higher oxygen conditions (D'Ambrosio and Harrison, 2021; Jensen et al., 2023; Peacock et al., 2021; Ray et al., 2024). Conversely, low oxygen creates conditions favourable for anaerobic methanogenesis (Audet et al., 2020; Bauduin et al., 2024). Together, these results suggest that seasonal shifts in biological activity, oxygen dynamics, and carbonate buffering interact to regulate CO₂ and CH₄ concentrations.

The SEMs also revealed several relationships between GHG concentrations, nutrients and sediment. For CO₂, we found a positive relationship between dissolved CO₂ and both TDC and sediment depth, suggesting that internal carbon loading and sediment respiration play significant roles in regulating CO₂ availability. This aligns with earlier findings that high organic matter loading, whether from allochthonous or autochthonous sources, contributes to elevated benthic respiration and the release

of CO₂ and dissolved carbon into the water column (Ray et al., 2023; Goeckner et al., 2022). Shallow ponds with a higher ratio of sediment surface area to water volume often exhibit elevated CO₂ concentrations, likely due to enhanced microbial activity at the sediment–water interface (Jensen et al., 2023). Importantly, this relationship was also reflected in the monthly trends: we observed the highest levels of TDC in June and August, which broadly coincide with elevated CO₂ concentrations and the highest CO₂ emissions in September. This supports the view that carbon accumulation over summer, followed by senescence-driven respiration in late summer and early autumn, may drive the release of CO₂ to the atmosphere (Ray et al., 2024).

In contrast, CO₂ fluxes were negatively related to TDP. Although nutrient enrichment is often associated with eutrophication and increased GHG emissions (Audet et al., 2020; Beaulieu et al., 2019; Malyan et al., 2022), it is well established that eutrophication can also drive CO₂ uptake in inland waters by stimulating autotrophic production (Balmer and Downing, 2011; Pacheco et al., 2014). In our ponds, TDP peaked in August, a month when both dissolved and emitted CO₂ were low, suggesting that phosphorus-driven algal or macrophyte growth may have temporarily enhanced photosynthetic CO₂ removal (Alleson et al., 2020). This pattern contributes to growing evidence that aquatic vegetation can act as a regulatory buffer, reducing CO₂ emissions through increased primary productivity (Bodmer et al., 2021; Ray et al., 2023).

For CH₄, we observed a negative relationship between TDN and dissolved CH₄ concentrations, consistent with known inhibitory effects of nitrogen compounds, particularly nitrate, on methanogenesis (Tugtas and Pavlostathis, 2008). Nitrate competes with methanogens for substrates and serves as an alternative electron acceptor, thus suppressing CH₄ production in sediments (Malyan et al., 2022). These dynamics mirror observations in other studies where nitrogen availability limits CH₄ production and promotes oxidation, especially under oxygenated conditions supported by macrophyte or microbial activity (Peacock et al., 2021; Jensen et al., 2023).

While previous work has shown that nutrient-rich systems can be strong CH₄ emitters, our findings highlight the context dependency of these relationships. Overall, the observed monthly variability and nutrient–GHG interactions suggest that internal biogeochemical feedbacks and temporal changes in nutrient cycling play a significant role in mediating GHG dynamics in agricultural ponds. These systems operate at the intersection of productivity and decomposition, with vegetation, nutrient supply, and sediment interactions all shaping their contribution to atmospheric carbon budgets (Lauerwald et al., 2023; Jensen et al., 2023).

4.2.3 Weather

We observed that increased atmospheric pressure was associated with higher concentrations of dissolved CO₂, a relationship that aligns with gas solubility principles. Elevated air pressure likely suppresses CO₂ degassing by increasing the equilibrium concentration of CO₂ in water, effectively “sealing” the water column (Wanninkhof et al., 2009). This physical constraint on diffusion has been reported in other freshwater systems, where short-term barometric pressure fluctuations influenced the vertical flux of gases (Cole and Caraco, 1998). However, a less intuitive pattern emerged for CH₄, where higher atmospheric pressure correlated with higher fluxes to the atmosphere. Since CH₄ is primarily emitted via ebullition, a gas-release process that occurs when methane bubbles in the sediment grow large enough to overcome the overlying sediment pressure and hydrostatic load (Zhao et al., 2022), this observation initially appears contradictory. Under higher pressure, ebullition should be suppressed as gas bubbles are more likely to remain dissolved or compressed within sediment porewater (Joyce and Jewell, 2003; Varadharajan and Hemond, 2012). One possible explanation is that atmospheric pressure may be acting as a proxy for other meteorological drivers, such as solar radiation, wind speed, or water temperature, that influence gas production, bubble formation, and release (Wik et al., 2014). For example, high-pressure systems are often associated with calmer, less windy conditions, which can reduce vertical mixing and stratification breakdown (Woolway et al., 2017), potentially affecting the timing and magnitude of GHG release from sediments.

4.2.4 Future research

While this study has shown that agricultural ponds emit significant amounts of CH₄ and CO₂, there is also evidence to suggest that ponds bury organic carbon at high rates (Taylor et al., 2019; Holgerson & Raymond, 2024). The net climate impact of these systems is therefore uncertain, as most current studies provide only snapshot measurements that do not capture the full life-cycle of ponds.

Evidence suggests that as ponds age, their sediment carbon stocks increase while per-area emissions may decline (Goeckner et al., 2022) but it’s currently uncertain if they shift to long term net carbon sinks. In many agricultural landscapes, ponds are periodically desilted or dredged to maintain water storage capacity, which removes accumulated organic carbon and prevents long-term burial. To fully understand the role of agricultural ponds in landscape carbon budgets more evidence is needed that accounts for both emissions and carbon stocks over longer time periods or using chrono sequence methods to identify how carbon dynamics change as ponds age.

The SEMs used in this study identified the key environmental drivers of GHG production and emission in agricultural ponds. By distinguishing between the factors influencing dissolved concentrations (in-pond production of GHG) and those affecting total fluxes (actual pond emissions), the SEMs revealed the relationships between nutrients, oxygen, pH, conductivity, and pond

morphology. These findings offer an empirical basis for improving our understanding of the mechanisms underpinning GHG dynamics in small aquatic systems. The next step is to translate these drivers into process-based models, which simulate causal relationships over time and enable the exploration of future climate scenarios.

Finally, it's important to note that this study has focussed on one ecosystem service from agricultural ponds. Whilst we have found the release of GHG to be a disservice it should not overshadow the combined positive benefits that agricultural ponds deliver to society through the many ecosystem services they provide.

5. Conclusion

This study quantified GHG emissions from agricultural ponds and investigated the environmental drivers that regulate their variability. We found that agricultural ponds are a consistent and substantial source of both CO₂ and CH₄, with monthly variation and strong sensitivity to environmental conditions (both within and beyond the pond). GHG emissions were shaped by a suite of physicochemical and physical characteristics. These findings highlight the importance of capturing sub-seasonal variability: studies that rely on a small number of samples collected during the summer season may overlook monthly fluctuations in GHG emissions, potentially leading to significant under or overestimations when scaling up to regional or global budgets.

5. Enhancing participation in agri-environmental schemes (AES): A Scottish case study

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Narrative author contribution statement

I led the design and development of this research, including formulating the research questions and methodological approach to assess participation in agri-environmental schemes. I designed the national survey instrument and coordinated engagement with farmers and advisers to support data collection. I was responsible for data curation, validation, and conducted the formal analysis of survey responses. I developed the analytical workflow and visualisations used to explore barriers to scheme uptake and participation. I led the interpretation of the findings, particularly in relation to policy design and the Scottish Government's proposed agricultural support framework, and wrote the manuscript, incorporating feedback from co-authors during revision.

Abstract

Scotland is undergoing a major period of agricultural reform, with a new agri-environmental scheme (AES) package aimed at positioning the country as a leader in sustainable and regenerative agriculture. For these reforms to succeed, farmer participation must be maximised through effective, accessible, and flexible scheme design.

A national survey of farmers and advisers was conducted to assess perceptions of current AES and identify barriers to uptake. AES were perceived to be financially risky, particularly due to uncertainty around application success and the potential consequences of non-compliance. Schemes were also seen as inflexible, with concerns centred on narrow application windows, limited control over contracted land, and rigid contract terms. While environmental awareness and social support were generally high, many respondents lacked clear information on which nature-based solutions (NbS) were appropriate for their farms. The application process was also perceived as overly complex, with poor feedback and limited communication from administering bodies.

The Scottish Government's proposed four-tier agricultural support framework offers a structured opportunity to address these challenges. Tier 2, which provides outcome-based and universally accessible payments, has strong potential to reduce perceived financial risk while offering greater flexibility through tailored options. To enhance awareness, information-sharing, and social acceptability, Tier 4 should support voluntary, farmer-led clusters with access to public and private advisory services. To improve application clarity, a streamlined online portal is recommended, modelled on England's Environmental Land Management (ELM) scheme, with clear, accessible guidance and built-in feedback mechanisms. These three recommendations align with the structure of the proposed support framework and directly address the key barriers identified by farmers and advisers. Their implementation would support the delivery of a more inclusive, practical, and effective AES capable of meeting Scotland's agricultural and environmental goals.

5.1 Introduction

Agri-environmental schemes (AES) are voluntary monetary based contracts offered to farmers and land managers; compensation is provided to cover incurred costs and productivity losses when specific management practices are implemented (Ait Sidhoum et al., 2023). They reward farmers for delivering positive environmental outcomes (Schaub et al., 2023) and are a key policy tool used by governments to steer farmers to manage their land in a nature and climate friendly manner (Eichhorn et al., 2023). AES therefore play an important role in supporting agricultural transition to a more sustainable system of food production, helping to address the 'triple challenge' of halting and reversing biodiversity loss, achieving net-zero greenhouse gas emissions, and producing food that is nutritious, affordable, and sustainable (WWF, 2023; Tyllianakis et al., 2023).

Prior to Brexit, Scottish AES were tied to the European common agricultural policy (CAP). In total, 75% of the payments delivered from the CAP were targeted for the conventionally focussed Pillar 1 of the CAP, with payments based on the area of land farmed (Pe'er et al., 2014). Only 25% of the CAP was paid through the competitive Pillar 2 rural development fund which supported AES (Science for Environment Policy, 2017). The CAP, and the AES associated with it, have been criticised for being ineffective in transitioning food production systems to more sustainable methods and meeting the requirements for the protection of nature and climate (Pe'er et al., 2017; Pe'er et al., 2020; Cullen et al., 2021; Hurly et al., 2022; Bateman & Balmford, 2018). However, Scotland's departure from the EU and the CAP offers an opportunity to create new agricultural policy (*Agriculture and Rural Communities (Scotland) Act 2024*). This policy shift is part of Scotland's future vision for the agricultural sector, where it aims to become a global leader in sustainable and regenerative agriculture with a support framework that delivers high quality food production, climate mitigation and adaptation, and nature restoration (Scottish Government, 2022a). Consequently, Scotland is undergoing an agricultural support reform process which began in 2024 and will transition through to 2027. This presents an opportunity to increase the portfolio and design of AES to increase their effectiveness for delivering environmental benefits (Schaub et al., 2023).

The new Scottish agricultural support system (Figure 21) is similar to the CAP in that it has multiple pillars or tiers with a tier 1 base payment to support continued farming. But there are marked differences with conditionalities attached to climate and nature for base payments. Furthermore, tier 2 is no longer competitive but is open to all and focussed on nature and climate. Tier 3 is competitive and in terms of environment is focussed on landscape scale nature and climate measures. The 4th tier of the proposed agriculture support package is for complementary support and is based around continuing professional development, advice, knowledge exchange and building

relationships with wider support networks. It will be delivered through the Farm Advisory Service (FAS) which is part of the wider agricultural knowledge and innovation system (AKIS) that links organisations, institutions, incentives and funding to address gaps in knowledge exchange and promotes uptake of learning activities.

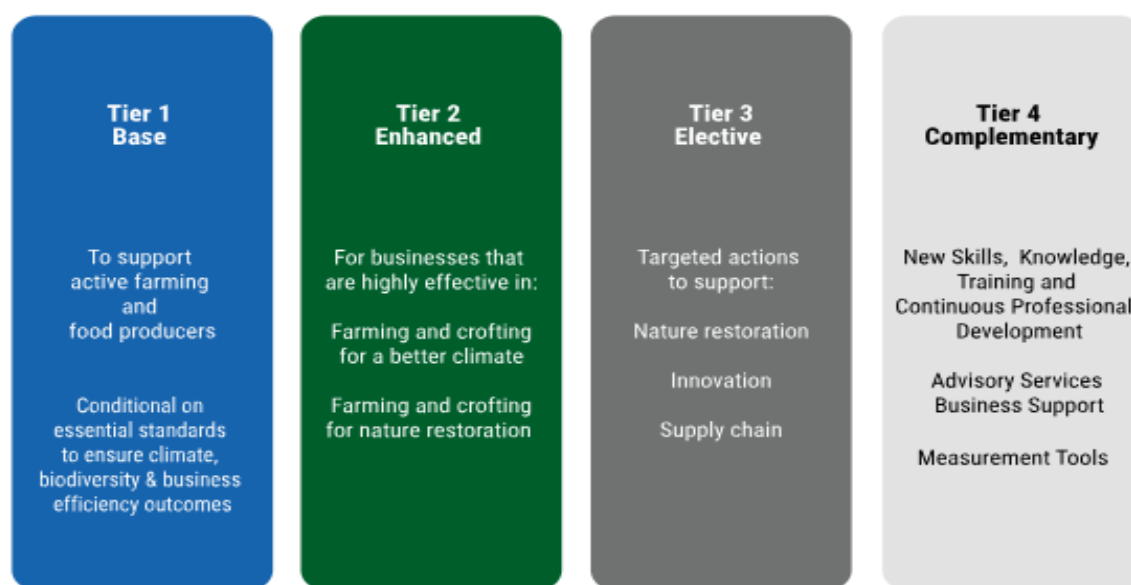


Figure 21: The 4 tiers associated with the new Scottish agricultural support systems.

This shift in the agricultural support system signals a change in mindset where farmers and land managers are now expected to provide a greater diversity of public goods (Cusworth & Dodsworth, 2021). Public goods are non-excludable (available to all) and non-rivalrous (one person using it does not reduce its availability to others) (Buchholz and Sandler., 2021). Examples include: flood resilience, access to green space, reduced greenhouse gas emissions, clean water, improved biodiversity etc. Securing public goods from agriculture will be essential in addressing the ‘triple challenge’ (Tylianakis et al., 2023) and transitioning to a future where nature is integrated into the food production system (Nature Friendly Farming Network, 2024).

To deliver these envisaged public goods from the new agricultural bill and agricultural support framework a significant proportion of Scottish farmers and land managers must adopt the new AES for it to be successful and effective (Canessa et al., 2024; Hurley et al., 2022). However, factors that influence AES uptake and adoption are complex (Tylianakis and Martin-Ortega., 2021; Klebl et al., 2024; Uthes and Matzdorf, 2013) and vary depending on the sociodemographic factors (Canessa et al., 2024). To understand the specific factors that are affecting uptake of AES in Scotland a survey was designed to evaluate how current Scottish AES are performing; specifically the agri-environment climate scheme (AECS), nature restoration fund (NRF) and the forestry grant.

The survey was designed around the primary research question: How do farm demographics, attitudinal factors, and perceived barriers influence the uptake of Agri-Environment Schemes in Scotland? To address this question, the survey investigates four key areas. First, it explores how farm demographics, including farm type, size, location, and the socio-demographic characteristics of farmers and advisers, affect participation in AES. Second, it investigates the interplay between attitudinal factors and perceived barriers to uptake. Attitudinal factors include environmental awareness (such as knowledge of environmental challenges and the perceived importance of stewardship), risk perception (including concerns about financial or regulatory uncertainty), and social acceptance (the extent to which AES participation is encouraged by peers or the wider farming community). These attitudes often shape or are shaped by perceived barriers, such as the inflexibility of scheme requirements (e.g., rigid contracts or restrictive land management rules), challenges in accessing relevant information (including guidance on eligibility, payments, and application processes), and the clarity and consistency of communication from government agencies. Finally, by analysing how these factors interact, the study identifies key issues influencing scheme uptake and uses these insights to develop evidence-based recommendations for policymakers aimed at improving participation in future Agri-Environment Schemes.

5.2 Methods

For access to the full survey instrument please see Appendix J

5.2.1 Survey Co-design process

Co-design is increasingly recognised as a valuable approach in agri-environmental research, helping to ensure that research tools are relevant, practical, and grounded in stakeholder experience (Reed, 2008; Blackstock et al., 2012). It involves actively involving end-users, such as farmers and advisers, in shaping the research process and outputs. This survey was co-designed with input from three farmers and two agricultural advisers from the Scottish Farm and Wildlife Advisory Group (ScotFWAG). The farmers were existing contacts from previous research, while the advisers were recruited specifically for this project.

The process began with informal discussions to identify key challenges and priorities related to Scotland's AES. These early conversations helped define the focus and structure of the survey. Based on these inputs, a pilot version of the survey was developed and shared with the same group. Farmers were visited in person on their farms to work through the pilot survey collaboratively, allowing them to ask questions and provide immediate, detailed feedback on content, structure, length, and clarity. Advisers were engaged remotely via telephone and email, they participated in initial discussions and later received the pilot and final survey drafts for comment. Their input

focused on technical language, accessibility of the questions, and practical concerns about scheme relevance. The advisers helped frame the terminology used in this survey to ensure the language was more accessible to farmers (Table 9). This iterative process allowed for several refinements, helping to ensure that the final survey was both robust and accessible. Given the ongoing reforms to Scottish AES, both farmers and advisers were motivated to contribute, which added credibility and stakeholder relevance to the design (Tyllianakis et al., 2023).

Table 9: Showing the interchangeable words used for the survey and project that farmers are familiar with.

Academic word	Words farmers can relate to
Agri-environmental schemes (AES)	Environment scheme
Nature based solutions (NbS)	Wildlife improvements
Public goods	Benefits

5.2.2 Survey structure and approach

The survey was comprised of three sections. The first section, titled ‘*Getting to know you and your farm*’, aimed to gather information about both the land being farmed and the characteristics of the farmer (Figure 22). This section followed a structure commonly used in previous UK farmer surveys (e.g. McMillan et al., 2019) and was designed to categorise respondents according to key farm demographics. These questions also allowed us to assess whether the sample was broadly representative of the wider farming population. Advisers who did not farm were not required to complete this section.



Figure 22: Farming background factors found to influence farmers decision on provision of 'public goods for public money' (McMillan et al., 2019)

The main component of the survey was called '*opinion questions*' and it examined what influences farmer's opinions of AES. A series of Likert scales were developed to facilitate this understanding. Six scales were created with several items in each scale (Table 10). The Likert scales were randomised to prevent any systematic bias of the survey design. The Likert scales and items were designed using the framework established by Canessa et al., (2024) which aims to measure the adoption of AES by covering a range of decision-making variables in a concise but thorough approach. The scales align with analogous research centred around farmers engagement with nature restoration (Jacob et al., 2023) and nature markets (Broadway initiative, 2023; GFI, 2023).

Table 10: Likert scales used within the survey. The construct measured is the "Likert scale" whilst the items to measure that specific construct are on the right.

Likert Scale	Items included in scale
Environmental Awareness	I am well-informed of the environmental issues in farming
	I know which wildlife improvements are most relevant for my local area
	I believe adding more wildlife improvements is beneficial
	I care for my local environment and feel a sense of stewardship
Social Acceptance	My community is supportive of environment schemes
	There is no stigma attached to adopting wildlife improvements in my community
	My family's approval motivates me to participate in environment schemes
	Positive influences from social media shape my opinion on environment schemes
Risks	Adding wildlife improvements is a financial risk
	It is risky applying for environment schemes because you might not be successful
	Environmental schemes are risky because of potential non-compliance
	I worry about accountability if issues arise with wildlife improvements.
	Environmental schemes have too much contractual risk
Inflexibility	There is the risk of having to maintain wildlife improvements after the contract ends
	Contract lengths are not flexible enough
	Adding wildlife improvements feel like an irreversible decision
	Adding wildlife improvements limits my control over land use
	The timeframe for applying to environment schemes is inconvenient
Information	The cost of getting information (advisors, surveys, databases) on what wildlife improvements are suitable is reasonable
	There is enough support to help me make decisions about environment schemes
	Farmer meetings help me get the information I need
	I get the information I need for environment schemes on time and it's adequate
Application clarity	Applying for environment schemes is simple
	Environment scheme guidance is easy to understand
	The language in environment schemes is easy to understand
	Communication with environment schemes is positive and cooperative
	Communication with environment schemes is prompt and clear.
	Case officers from environment schemes have a good understanding of farm requirements and operation
	Feedback from environment schemes is clear and constructive

Likert-scale item responses were scored on a 0–5 scale (0 = Strongly disagree to 5 = Strongly agree) and aggregated to form composite measures for each attitudinal construct. The mean of item responses was calculated for each respondent, consistent with established practice when Likert data are treated as approximating interval-level properties for parametric analysis (Norman, 2010). This method preserves response variability, enables group comparisons, and supports further statistical procedures such as regression and correlation. Comparison of mean and median values revealed differences of less than 0.1, supporting the use of the mean as a robust summary statistic. Composite scores enhanced construct-level interpretation and reliability, enabling more stable and interpretable measurement of latent attitudes across items (DeVellis, 2017).

5.2.3 Survey distribution and uptake

The survey targeted farmers and agricultural advisers in Scotland, recognising that AES differ across the devolved nations of the UK. The study aimed to capture a broad range of perspectives from those involved in land management and farming, including full-time and part-time farmers, crofters, tenant farmers, owner-occupiers, and professional advisers. This diversity of respondents ensures that a wide spectrum of experiences, roles, and responsibilities in relation to AES participation are represented (Rey-Valette et al., 2017).

The survey was disseminated online via farming networks and newsletters (Table 11). The approach also recognised, however, that a substantial proportion of the farming demographic are older and that this might present barriers with regard to their engagement with online surveys. This was addressed by having paper copies of the survey at locations where the farming community converge such as auctions, markets, conferences, farmer meets, agricultural shows etc.

Table 11: Organisations contacted that disseminated the farmer survey and their method of dissemination.

Organisation	Format
Northwoods partnership	Newsletter
ScotFWAG	Email and word of mouth
NFFN	Newsletter and social media
SRUC/SAC	Email and word of mouth
Scottish crofting federation	Newsletter
RSPB	Email and word of mouth
Scottish wildlife trust	Email and word of mouth
Loch Lomond and Trossachs national park authority	Social media
Cairngorms national park authority	Email and word of mouth
Forth rivers trust	Newsletter and social media
NFUS	Newsletter and email
Pasture for life	Members forum
Scottish farmers forum	Social media
Soil association	Newsletter
AFN Network+ (Agri-food for Net Zero Network)	Newsletter
BASE-UK	Newsletter
AHDB	Email and word of mouth

5.2.4 Statistical analysis

A variety of data types were collected through the survey, including categorical, ordinal (e.g. Likert-scale items), continuous, and count data (Table 12). Given the predominance of non-normally distributed, ordinal, and count variables, non-parametric methods were primarily employed to ensure robust analysis without relying on assumptions of normality or homogeneity of variance. Associations between two categorical variables were analysed using the chi-square test of independence, used for testing relationships between nominal variables (Agresti, 2018). When the independent variable was categorical and the dependent variable was continuous, ordinal, or count-based, the Kruskal-Wallis test was applied. This non-parametric alternative to one-way ANOVA is appropriate for comparing multiple groups when data do not meet parametric assumptions (Sheskin, 2003). To assess relationships between variables where at least one was ordinal or count-based, Spearman's rank correlation coefficient was used. This test is suitable for analysing non-parametric

data and does not require the assumption of normally distributed variables, making it appropriate for survey responses such as Likert-scale items and other ranked or count-based measures (Hauke & Kossowski, 2011). Relationships between continuous-continuous or count-count variables were not examined, as they were not relevant to the study's research questions.

Table 12: Statistical test used, depending on the data type combination.

	Categorical	Continuous	Likert (Ordinal)	Count
Categorical	Chi-square + Cramer's V	Kruskal-Wallis	Kruskal-Wallis	Kruskal-Wallis
Continuous	-	N/A	Spearman's Rank	Spearman's Rank
Likert (Ordinal)	-	-	-	Spearman's Rank
Count	-	-	-	N/A

To assess the internal consistency of the Likert scales, Cronbach's alpha was calculated for each scale. Cronbach's alpha is a widely used reliability coefficient, used to determine whether a group of related items reliably measures a single underlying construct. It is calculated based on the average inter-item correlation among a set of items within a scale. It considers both the number of items in the scale and the average covariance between item pairs (Tavakol and Dennick, 2011).

In this study, each Likert scale reflected a distinct attitudinal theme relevant to AES uptake, such as risk perception, environmental awareness, information availability and application clarity. Each scale included multiple items, individual statements rated by respondents on a 5-point Likert scale, designed to capture different dimensions of that theme. For example, the risk perception scale included items about AES competitiveness, non-compliance concerns, and financial uncertainty.

Cronbach's alpha produces a coefficient between 0 and 1, where higher values indicate stronger internal consistency among the items within a scale. A high alpha score suggests that the items are closely related and likely measuring the same underlying idea. For instance, if respondents who agree that "Environmental schemes have too much contractual risk" also tend to agree that "Environmental schemes are risky because of potential non-compliance," this indicates a consistent pattern across the risk-related items. Conversely, a low alpha would suggest that the items are not well-aligned or may reflect multiple unrelated ideas. While interpretation thresholds can vary, a commonly accepted threshold is 0.7 or above for acceptable reliability, although values above 0.6 can be considered acceptable in exploratory research (Tavakol and Dennick, 2011).

Applying Cronbach's alpha in this study served two key purposes. First, it provided a statistical check to confirm that the items grouped within each scale were conceptually aligned and measuring a coherent construct. This is particularly important for attitudinal data, where consistency among responses strengthens the validity of interpreting scale scores. Second, it supported the rigour of the co-designed survey. Although the items were developed through consultation with farmers and advisers, and informed by previous studies, reliability testing ensured that the resulting scales were not only contextually relevant but also methodologically robust. This improved confidence on the use of these scales in subsequent analysis that explored what influences farmer engagement with AES.

5.3 Results

This section presents the findings from the survey, organised into four parts. First, a descriptive overview of respondents and their land is provided, covering key farm demographics and respondent characteristics. Second, the results from the Likert-scale items are presented, offering insights into attitudes, perceptions, and barriers related to AES participation. Third, internal consistency of the Likert-scale groupings is assessed using Cronbach's alpha. Finally, statistical tests of association explore relationships between farm and demographic variables and the attitudinal and barrier-related responses, as well as associations within individual sections of the survey.

5.3.1 Getting to know you and your farm

In total, 80 survey responses were received: 38 from farmers; 25 from advisers; and 17 from advisers who also farm. The majority of respondents were university educated (80%). Most respondents were male (67%). In terms of location, most responses were attributed to the east central region (23%) while 13% of responses indicated 'Scotland', which represents advisers who work across multiple regions (Figure 23).

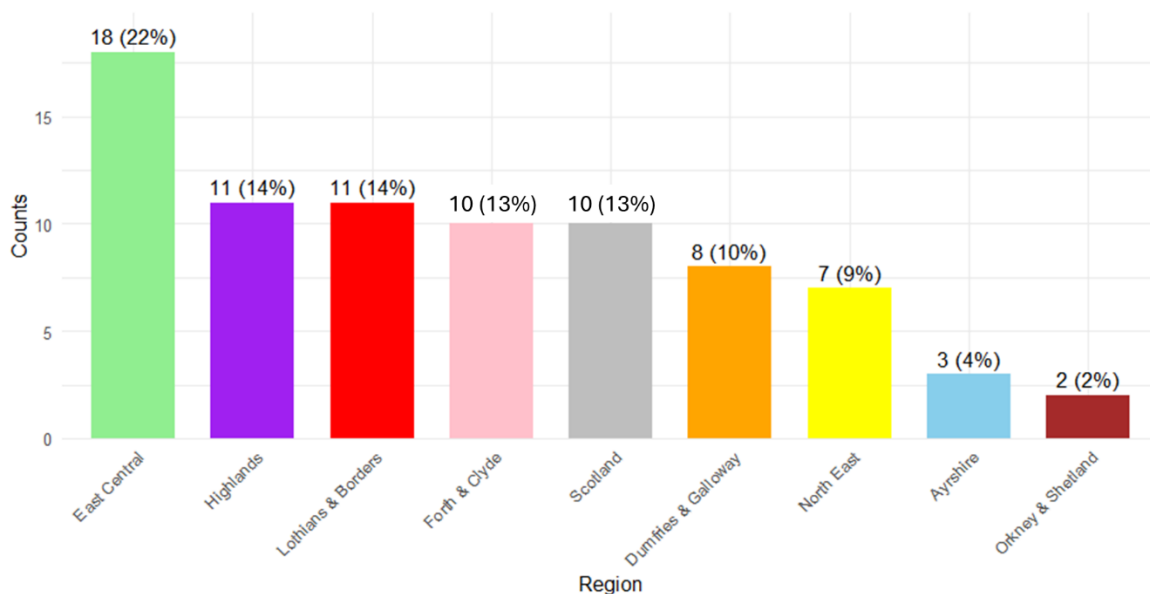


Figure 23: Where respondents primarily farm or advise, bar plot showing count data along the x axis and percentages for each region.

The most common farming activity, reported by the majority of farm enterprises, was livestock agriculture (58%), followed by crop production (27%), with woodland management being the least common (15%). Overall, 63% of farmers stated that they farmed on less favoured areas (LFA) status land. With regard to ownership, 71% of the farmers owned their farms, 15% were tenants and 15% had mixed ownership/tenureship. The median farm size was 140 hectares with a few very large (i.e. > 1000 ha) land holdings responding. Within the survey responses, 47% of farmers indicated that they are currently engaged in an AES. Most farmers and advisors who farm accommodated a few wildlife improvements (NbS) on their land, with several respondents having >5 NbS on their land (Figure 24a). The most common wildlife improvement on farms was grassland managed for nature followed woodland planting and hedgerow planting/restoration. The least common wildlife improvements were those that required large investments of resources such as restoring natural rivers and the construction of rural sustainable drainage systems and also those specific to certain land types (i.e., moorland management) (Figure 24b).

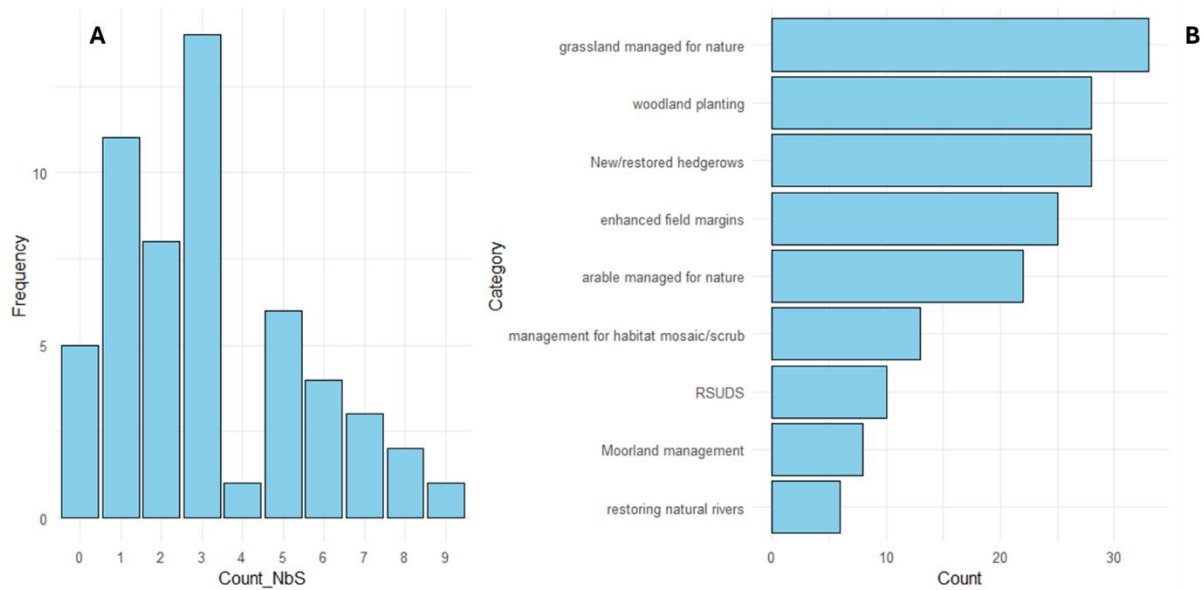


Figure 24: A shows count data for the number of NbS on each farm. B shows the count data for each type of wildlife improvement on the farms.

The final question of this section asked farmers ‘How important are the following reasons for your decision to farm’ (Figure 25). All the options were important for farmers, though some more so than others. Stewardship of the land emerged as a key theme, with ‘looking after the countryside’ and ‘keeping the land in good heart’ rated as very important by the majority of respondents. This aligns with the commonly discussed view of farming as more than just an occupation. While for some, it may reflect a lifestyle preference, given that over half of respondents found ‘enjoying the farming way of life’ to be very important, for others, it may be more strongly shaped by intergenerational succession and family obligation rather than a freely made career choice.

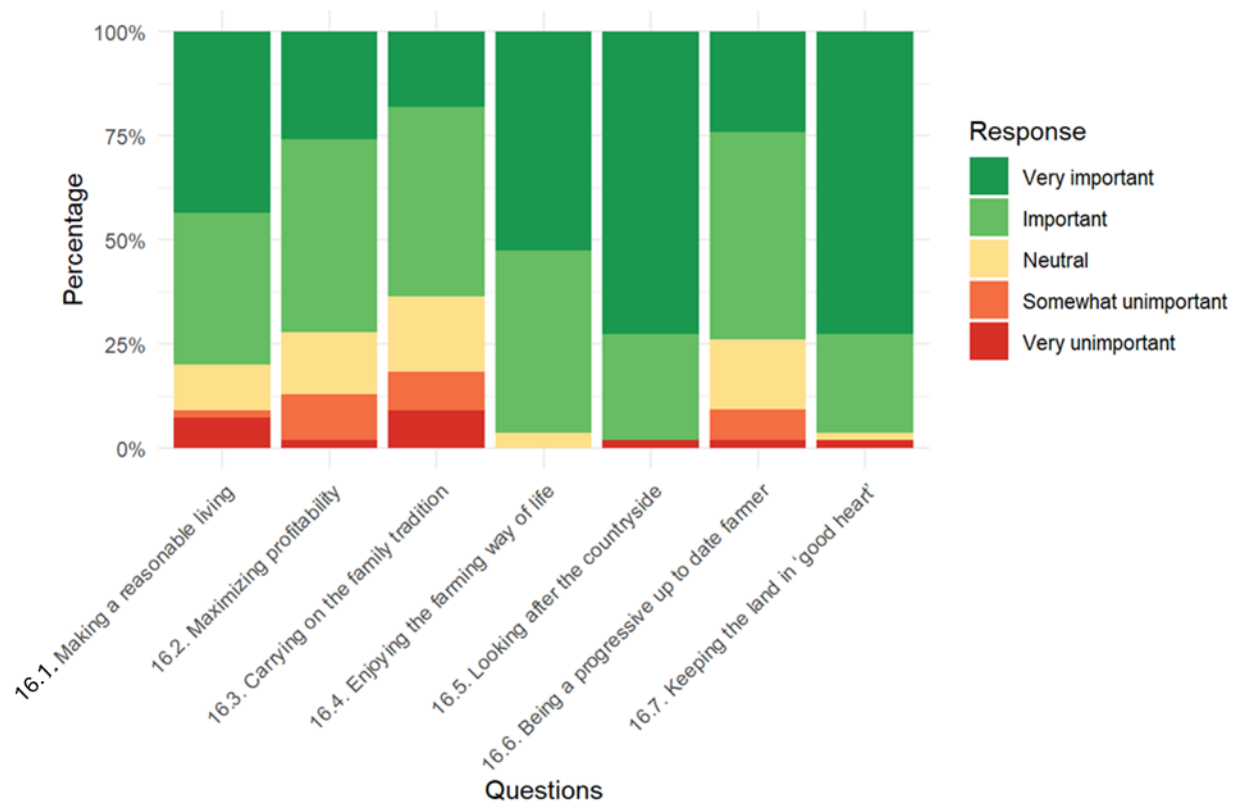


Figure 25: Likert scales showing how important various reasons are for farming.

5.3.2 Likert Scales

A series of Likert scale responses provided an overview of the attitudinal factors and perceived barriers affecting AES (Figure 26). Respondents reported high levels of self-perceived awareness of environmental issues, with over 84% agreeing or strongly agreeing with each of the four statements in the environmental awareness scale (Figure 26a). These statements covered topics such as being informed about environmental issues in farming, believing in the benefits of adding wildlife improvements, caring for the local environment, and feeling a sense of environmental stewardship. The highest agreement was observed for items relating to environmental care and stewardship, with 93% of respondents indicating agreement or strong agreement. However, one item stood out: when asked whether respondents knew which wildlife improvements were most relevant for their local area, only 28% strongly agreed, significantly lower than the average of 59% for the other three items (Fig 26a).

Responses to the scale measuring social acceptance of AES were more varied than those for environmental awareness, with a notable proportion of participants choosing neutral responses, averaging 31% across all 4 items (Fig. 26b). Overall, respondents tended to agree that their local community is supportive of environmental schemes, that there is no stigma associated with

adopting wildlife improvements, and that family approval acts as a motivating factor. For these three items, the average proportion selecting “agree” or “strongly agree” was 42%, compared to 15% who disagreed or strongly disagreed. However, the item addressing the influence of social media on opinions about AES showed a much more divided pattern: 22% of respondents agreed and 22% disagreed, with a higher proportion strongly disagreeing (11%) than strongly agreeing (5%).

The Risks of AES scale measured how strongly respondents associated AES participation with different types of risk. For all five items in the scale, selecting “agree” or “strongly agree” indicated a higher level of concern or perceived risk, while “disagree” or “strongly disagree” reflected lower concern. On average, just over half of respondents (53%) perceived an overall moderate to high risk associated with AES, with 15% strongly agreeing and 38% agreeing. In contrast, only 19% expressed low concern (16% disagreeing, 3% strongly disagreeing), while 24% gave neutral responses (Fig. 26c). The highest perceived risks were related to uncertainty and potential consequences: many respondents agreed that applying for AES is risky because success is not guaranteed, that non-compliance could lead to penalties, and that they could be held personally accountable if issues arose from wildlife improvements. A second tier of concerns focused on contractual complexity and the financial risks associated with implementing wildlife measures.

Responses to the inflexibility scale, where agreement indicated greater perceived inflexibility and disagreement indicated perceived flexibility, suggest that many participants viewed AES as rigid in their structure. When averaging across four of the five items in the scale, excluding the item about whether adding wildlife improvements feels irreversible, 43% of respondents agreed or strongly agreed, while only 14% disagreed or strongly disagreed (Fig. 26d). The strongest concerns centred on three key areas: inconvenient application timeframes (33% agree/strongly agree), inflexible contract lengths (54%), and the risk of having to maintain wildlife improvements after contracts end (47%). These items reflect a clear sense of procedural and long-term constraint. Perceptions were more mixed regarding loss of control over land use, with 34% agreeing but 27% disagreeing, indicating a more divided view. The clearest departure from this overall pattern was in response to the statement that “adding wildlife improvements feels like an irreversible decision”, here, 43% of respondents disagreed or strongly disagreed, making it the only item in the scale where rejection outweighed agreement.

The information accessibility scale produced a mixed pattern of responses, reflecting neither strong satisfaction nor widespread dissatisfaction among participants (Fig. 26e). Averaging across the first

three items, concerning the cost of obtaining information (e.g., from advisors, surveys, or databases), the availability of support for decision-making, and whether information is received in time and is adequate, 35% of respondents agreed or strongly agreed, 23% were neutral, and 30% disagreed or strongly disagreed. These results suggest that while a portion of respondents found information provision around AES to be reasonable and supportive, a comparable share felt that it fell short in terms of timeliness, cost-effectiveness, or adequacy. In contrast, responses to the final item “Farmer meetings help me get the information I need” were notably more positive: 58% agreed or strongly agreed, while only 19% disagreed or strongly disagreed.

Responses to the application clarity scale showed that many participants found the process of engaging with AES confusing or difficult. In this scale, disagreement with a statement reflects a perception of poor clarity. On average, 46% of respondents disagreed or strongly disagreed, while only 14% agreed or strongly agreed, indicating generally low satisfaction with how clear and accessible the application process is (Fig. 26f). The most negatively rated aspects were the simplicity of applying and the clarity of guidance documents. For these two items, 72% and 59% of respondents, respectively, disagreed or strongly disagreed that the process was straightforward. Other items in the scale received slightly less negative but still concerning responses. For example, 47% of respondents disagreed that the language used in AES materials is easy to understand. Communication was also a common source of frustration: 42% felt that communication with schemes is not prompt or clear, and 30% disagreed that it is positive and cooperative. Similarly, 40% said that feedback from AES is not clear or constructive, and 31% disagreed that case officers understand farming needs and operations. Many of these communication-related items also had relatively high “don’t know/prefer not to say” response rates, averaging 22%. This likely reflects the fact that fewer than half of respondents were actively involved in an AES, limiting their direct experience with these elements.



Figure 26: Responses for each of the Likert scales from the survey respondents with legend beneath.

5.3.3 Cronbach Alpha

To test the internal consistency of the Likert scale items, Cronbach's alpha scores were calculated for each of the attitudinal factors and perceived barriers. These reliability scores indicate how well the items within each Likert scale measured a common underlying concept. Higher values suggest greater internal consistency. Social acceptance and Information scales have a moderate score (>0.6) whilst environmental awareness and inflexibility have high scores (>0.7) whilst application clarity has a very high score (>0.8) (Table 13).

Table 13: Showing Likert scales with associated Cronbach's Alpha scores.

Likert scale	Cronbach Alpha score	Number of Items
Environmental awareness	0.74	4
Social acceptance	0.63	4
Risk	0.76	5
Inflexibility	0.72	5
Information	0.64	4
Application clarity	0.86	7

During the data cleaning stage, two items were reassigned from the Information scale to the Application Clarity scale. The items; '*Applying for environment schemes is simple*' and '*Environment scheme guidance is easy to understand*', demonstrated both low corrected item-total correlations ($r_{\text{cor}} < 0.4$) and low item-total correlations ($r_{\text{drop}} < 0.3$) when included in the Information scale. These values indicated that the items were not strongly associated with the other items in that scale and were therefore not internally consistent with the construct being measured.

In contrast, the other items in the 'Information' scale had acceptable item-total correlation values ($r_{\text{drop}} > 0.3$), suggesting they were conceptually and statistically coherent as a group. The two items selected for reassignment differed substantively in focus, addressing perceived clarity and ease of scheme application, rather than the accessibility or quality of informational resources. Moving them to the Application Clarity scale, which better matched their content, led to stronger internal consistency in both scales, helping ensure that each set of items measured a single, coherent idea.

5.3.4 Relationships Between Respondent Profiles, Perceived Barriers, and Attitudes Toward AES

Associations between social acceptance of AES and farm(er) characteristics

Statistical tests examining associations between social acceptance of AES and farm(er) characteristics revealed several significant patterns. Social acceptance was significantly lower among farmers compared to advisers ($P = 0.007$). Among farmers, higher levels of perceived social acceptance were associated with a greater number NbS implemented on their farms ($P = 0.03$). In addition, tenant farmers reported lower social acceptance of AES than owner-occupiers ($P = 0.026$). A positive correlation was observed between high social acceptance and the presence of more NbS on farms (Figure 27a-c).

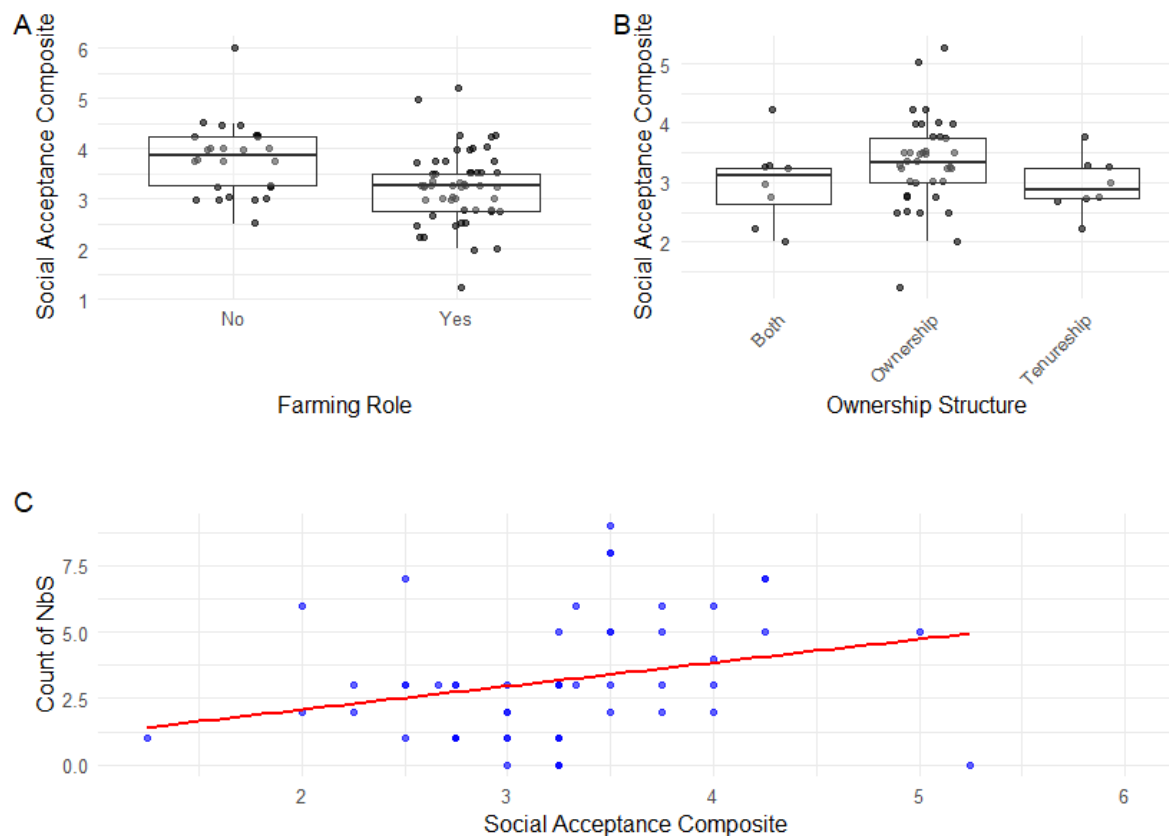


Figure 27: Plots associated with the social acceptance of AES. Top left = Social acceptability of AES vs whether the respondent farms or not. Top right = Social acceptability of AES vs whether the ownership structure of the farm. Bottom = Scatter plot showing social acceptability of AES vs the number of NbS on the farm.

Associations between environmental awareness and farm(er) characteristics

A series of farm and respondent characteristics were examined for their association with levels of perceived environmental awareness. Respondents with a university-level education reported significantly higher environmental awareness compared to those educated to high school or college level ($P = 0.001$). Environmental awareness was also lower among farmers and advisers who actively farm, compared to advisers who do not farm ($P = 0.03$). This may relate to differences in educational background (Figure 28C). The heatmap in Figure 28C presents the association between education level and whether respondents currently farm or manage agricultural land. Farmers and farming advisers were significantly less likely to hold a university degree than non-farming respondents, and more likely to report college or high school as their highest level of education ($P = 0.0002$, Cramér's $V = 0.36$). The coloured residuals indicate where the observed frequencies differ most from what would be expected by chance: blue cells show fewer responses than expected, and red cells show more. Among farmers, higher levels of environmental awareness were associated with a greater number of nature-based solutions (NbS) present on their farms ($P=0.034$) (Figure 28a-d).

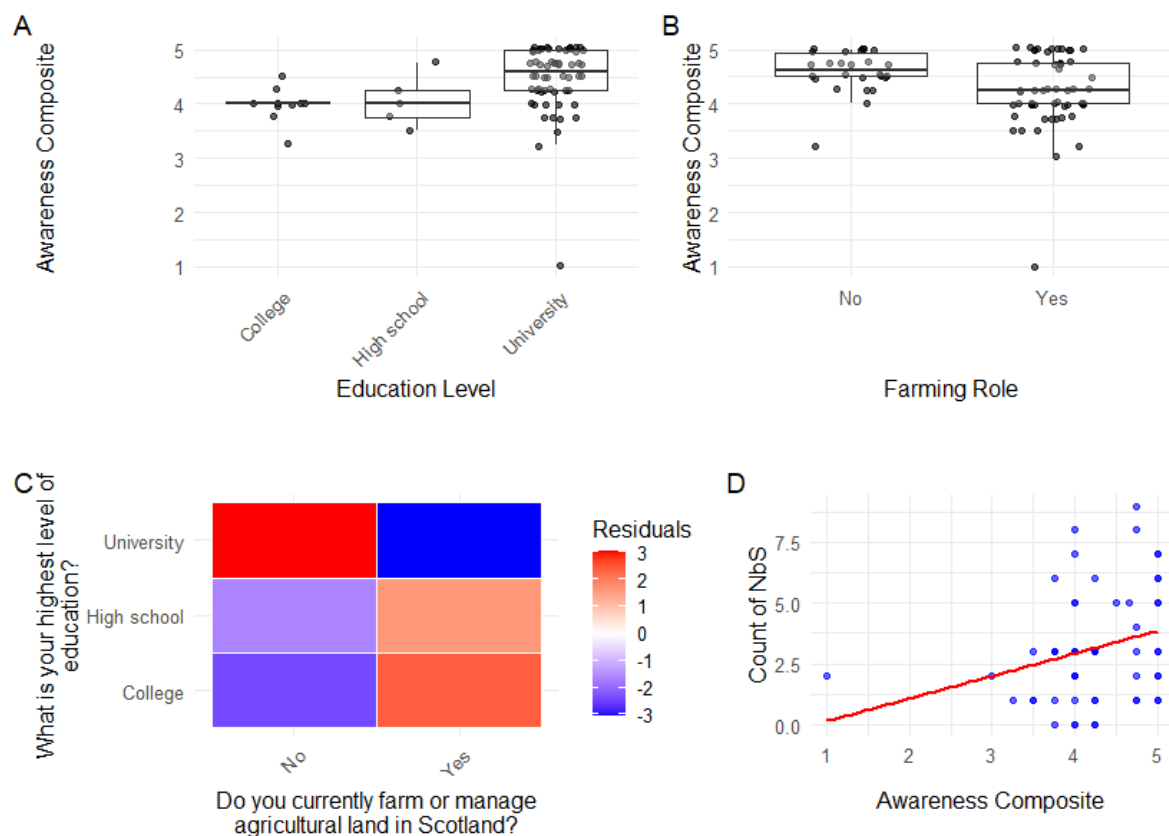


Figure 28: Plots associated with awareness of environmental issues. A= box plot showing education level against environmental awareness. B = Environmental awareness against whether the respondent farms or not. C = Heatmap showing level of education vs whether the respondent farms or not. D = Scatterplot showing number of NbS on a farm on the y axis vs environmental awareness on the x axis.

Associations Between Perceived Inflexibility and risk of AES and Farm(er) Characteristics

Perceptions of inflexibility in AES varied across different respondent groups. Advisers perceived AES as more inflexible than farmers, indicating a higher level of concern about scheme rigidity among advisers ($P = 0.004$). Among farmers, those who were currently engaged in AES perceived greater inflexibility than those not participating in a scheme at the time of the survey ($P = 0.02$). Established farmers also viewed AES as more inflexible compared to new entrants to farming ($P = 0.003$). In addition, farmers who perceived AES as more inflexible tended to have fewer nature-based solutions (NbS) implemented on their farms. Similarly, Farmers who perceived AES to be highly risky had less NbS on their farms ($P = 0.04$) (Figure 29a-e).

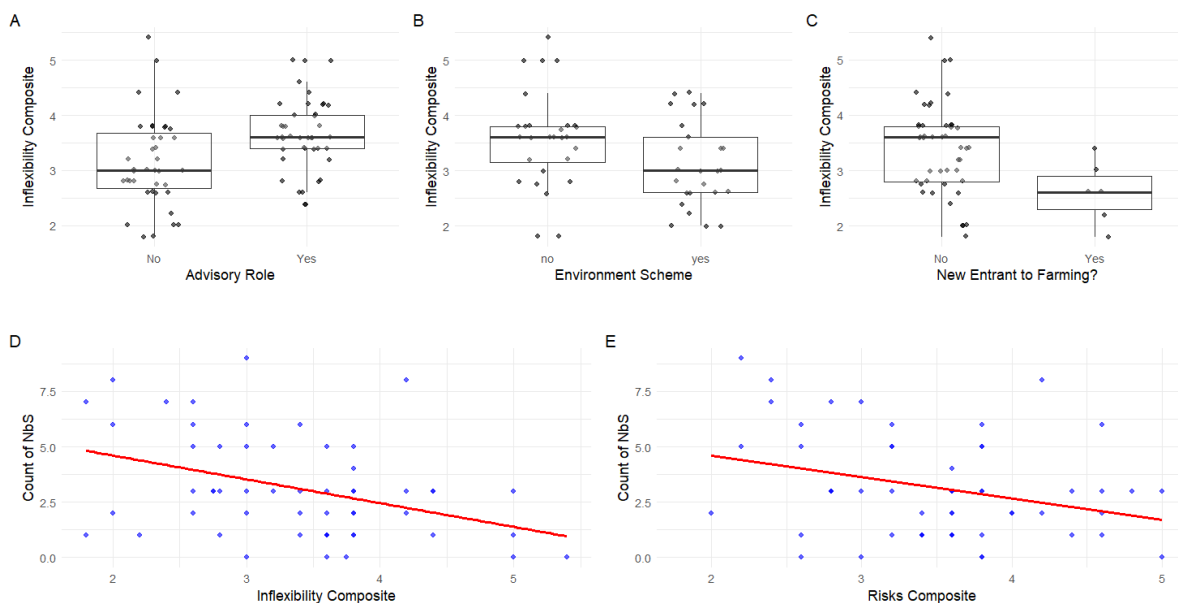


Figure 29: A = Boxplot showing Inflexibility of AES vs whether the respondent work as an adviser, B = Boxplot showing Inflexibility of AES vs whether the farm is engaged in an environment scheme. C = Boxplot showing Inflexibility of AES vs whether the farmer is a new entrant or not. D = Scatter plot showing flexibility of AES against number of NbS on a farm. E= Scatter plot showing how perceived risk of AES changes with the count of Nature based solutions on a farm.

Associations Between Perceived Application Clarity of AES and Farm(er) Characteristics

Perceptions of AES application clarity differed across respondent groups. Advisers reported lower application clarity than farmers, suggesting they view the process as more complex or difficult ($P = 0.008$). Among farmers, however, those currently participating in AES perceived application clarity to be lower than those not involved in a scheme ($P = 0.02$). In addition, farmers with larger land holdings reported lower levels of application clarity compared to those with smaller holdings. Farmers already engaged in AES had more NbS on their farms ($P = 0.001$, Chi squared = 10.83) (Figure 30a-c).

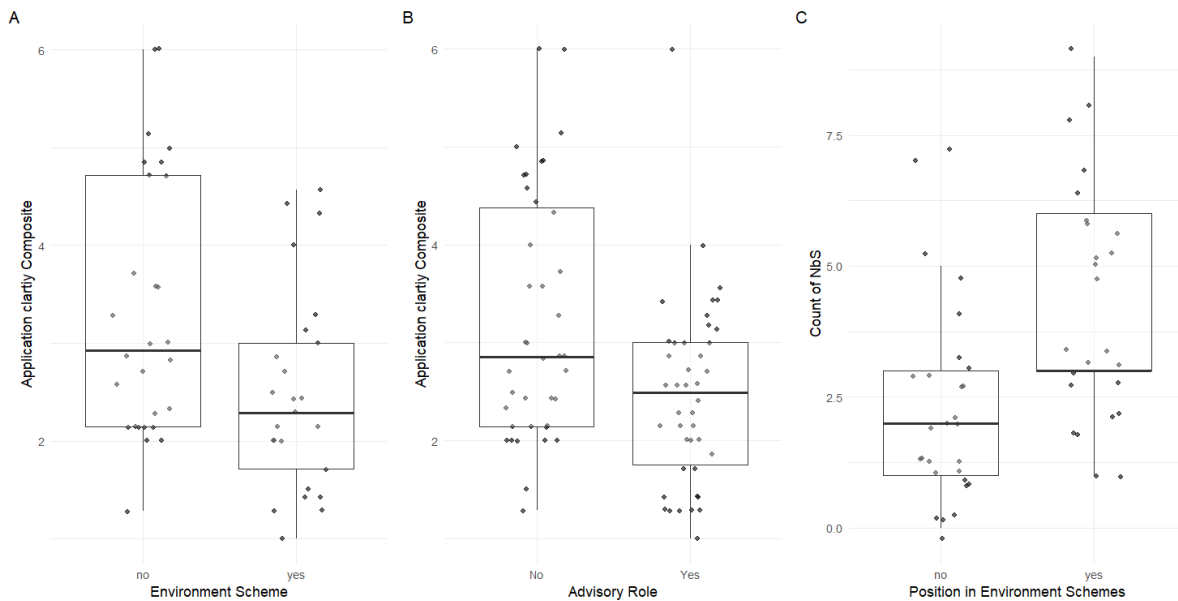


Figure 30: A = Boxplot showing application clarity vs whether the farmer is engaged in an environment scheme. B = Boxplot showing application clarity vs whether the respondent works as an adviser. C = Boxplot showing how the count of NbS on a farm differs with whether the farmer is engaged in an environment scheme or not.

5.4 Discussion

Our survey of farmers and advisers provides new insight into the social and structural factors that shape engagement with AES and the adoption of NbS on farms. By examining both attitudinal dimensions, such as environmental awareness and social acceptance, and perceived barriers including risk, inflexibility, and application clarity, this study offers a more comprehensive understanding of the behavioural context in which AES operate. Importantly, we found that perceptions of AES vary systematically by role, experience, and farm characteristics, and that these perceptions are closely linked to on-farm environmental action. These findings address a critical knowledge gap in understanding the non-financial drivers of AES participation. As agricultural policy in Scotland undergoes a major transition under the Agriculture and Rural Communities (Scotland) Act 2024, and with changes to the Scottish agricultural support system on the horizon, our results underscore the importance of designing schemes that are not only technically robust, but also socially acceptable, accessible, and clearly communicated. Below, we reflect on the implications of these findings in the context of ongoing policy reform.

5.4.1 Social acceptability, information and education

The social acceptability of AES plays an important role in farmers' decision-making (Dessart et al., 2019) and can significantly influence their willingness to engage with and participate in AES (Eichhorn et al., 2024; Lastra-Bravo et al., 2015). This may be why the farmers in this survey who had greater social acceptance of AES had more NbS on their farms. Respondents generally perceived the social acceptance of AES to be high, with both family and community support viewed as important for promoting engagement with AES. This is encouraging as the opinions and experiences of farmers' peers and local community affect their decision-making (Burton and Paragahawewa, 2011; Emery and Franks, 2012) and influence their willingness to adopt AES (Defrancesco et al., 2008). However, our findings also revealed a divide: tenant farmers perceived AES to be less socially accepted than owner-occupiers. This is probably due to tenant farmers often facing reduced autonomy over land use decisions, limited lease durations, and uncertainty about long-term land access, which can restrict their capacity or willingness to engage with AES (Mills et al., 2019; Mills et al., 2021).

Enhancing social acceptance among farmers could be a valuable strategy for increasing participation in AES. While family and community support were widely recognised as positive drivers of engagement in this study, responses to social media as a source of influence were more polarised. Although social media platforms have the potential to foster pro-environmental attitudes and facilitate knowledge exchange among farmers (Zhou et al., 2021), they can also introduce conflicting narratives, misinformation, or social comparison pressures that reduce trust and motivation (Zhang et al., 2021). Nevertheless, social media may still support engagement with AES by enabling farmers to share experiences, build digital credibility, and influence peers; functions that could complement more traditional, place-based forms of trust-building and communication (Soulignac et al., 2025).

An awareness of environmental issues is an important component in the adoption of AES (Klebl et al., 2024; Lastra-Bravo et al., 2015) and is closely linked to ideas of farm stewardship which has been shown to encourage sustainable farm practices and adoption of AES (Tankosić et al., 2023). The respondents in this survey already care for their local environment and feel a sense of stewardship, which is an important step in engaging farmers with AES (Dessart et al., 2019; Canessa et al., 2024). Furthermore, respondents felt they were well informed about the environmental issues that cut across farming and believed that adding NbS was beneficial. This combination of environmental awareness and stewardship may explain the observed correlation between farmers with high environmental awareness and the increasing amount of NbS on their farms. However, when respondents were asked if they knew which NbS were most relevant for their local

area there was less certainty. This may be tied to disparities in the level of education and environmental awareness between those respondents who farmed and those who do not. To address this gap, educating farmers about NbS may offer an effective approach for promoting the engagement of AES and increasing the provision of public goods (Zindler et al., 2024). The majority of research (Wilson and Hart, 2000; Okumah et al., 2021; Dessart et al., 2019) shows a positive link between environmental awareness and AES participation. However, some studies, like Schaub et al. (2023), found no relationship, which may be because, as shown in a meta-analysis, payments can reduce farmers' personal motivation to protect the environment, leading some who care about it to choose not to take part.

The better informed a farmer is, the more likely they are to participate in AES (Lastra-Bravo et al., 2015) and to remain actively involved with AES (Morris, 2006) which will likely lead to changes in farm systems and practices (Pannell et al., 2006). Therefore, investing in farmer education can also improve access to information, which was identified in this study as an area of current AES design that could be improved. Providing this education via training and advice can enhance farmer's perception and participation of AES (Schaub et al., 2023; Tankosić et al., 2023). Analogous studies have found that this information is best received when it includes practical guidance on implementation and is tailored to the farmer (Klebl et al., 2024). The level of information a farmer has about the structure, requirements, and outcomes of AES can significantly influence their perceptions of risk and uncertainty, particularly around scheme complexity, compliance obligations, and the likelihood of success (Whitten et al., 2013; Dessart et al., 2019; Canessa et al., 2024).

5.4.2 Risk and inflexibility

Farm businesses face a wide range of risks, from market volatility to weather and climate change, which naturally makes farmers risk-averse (Vollenweider et al., 2011). In our survey, farmers perceived the overall risk of applying for and participating in AES to be high, viewing it as another potential threat to their farm business. This may help explain why those who saw AES as more risky tended to have fewer nature-based solutions (NbS) on their farms. Reducing this perceived risk is therefore crucial for increasing farmer participation in AES (Eichhorn et al., 2024). The reason for this high perceived risk is that current AES in Scotland are competitive meaning applicants run the risk of putting time and resources into an application and then failing to be accepted. If the farmer is accepted onto the AES then they face a mixture of regulatory and contractual responsibilities, which results in uncertainty around payments (Cardwell, 2023). This is likely why the two highest perceived risks from the respondents in this survey were relating to application success uncertainty and non-compliance. This combination of insecurity before and during AES implementation can negatively affect AES participation (Bartkowski et al., 2019). The respondents in this survey also perceived AES

to be too financially risky, which plays an important role in the decision-making process of engaging with AES (Sattler and Nagel, 2010; Riley, 2016; Zindler et al., 2024).

Risk and inflexibility of AES are strongly linked, e.g., AES contracts that are flexible may increase participation of more risk-averse farmers (Schaub et al., 2023). Building flexibility within AES has been identified as key to securing benefits for farm businesses and the environment at both the farm and landscape scale (Nature Friendly Farming Network, 2024). Respondents believed current AES are too inflexible with the main issues stemming from the timeframe for applying, control of land, and contracts, all of which negatively affect AES participation (Mettepenningen et al., 2013). Again, we see a disparity between respondents who farm and those who do not with higher perceived inflexibility for farmers. Furthermore, we see that those established farmers and those currently engaged in AES had perceived AES to be less flexible than new entrants and those not engaged. This suggests that as farmers progress through their careers and engage with AES they increasingly perceive them to be less flexible. This is important because these negative experiences erode institutional credibility and affect AES participation (Sutherland et al., 2013).

Farmers prefer AES and their measures to be less restrictive and more flexible (Ruto and Garrod, 2009) which could be why those farmers who perceived Inflexibility to be high had less NbS on their farms. Providing more flexibility of the AES through a range of different levels of AES engagement and associated payment levels may encourage the participation of AES by allowing farmers to take on what they feel comfortable with (Lastra-Bravo et al., 2015). However, tailoring each AES down to the individual farm is not feasible and so there needs to be a broad enough range of flexible measures that encompass the different farming landscapes and businesses found in Scotland (van der Ploeg and Ventura 2014; Klebl et al., 2024)

5.4.3 Application clarity

The clarity of the AES application process was perceived by respondents to be poor and there seems to be two compounding reasons for this. Firstly, respondents find the application process difficult to understand, complex and inaccessible, all of which act as barriers to participation in AES (Birge et al., 2017; Tankosić et al., 2023 ;Canessa et al., 2024). Farmers are more likely to participate in AES when the process is easy and does not 'cost' them much in terms of time and resources (Lastra-Bravo et al., 2015). The current AES process has a heavy bureaucratic load which acts as a barrier to participation (Ruto and Garrod, G., 2009; Bartkowski et al., 2023). Simplifying and making the application guidance more accessible will help to remove this barrier.

Secondly, as reflected in the survey responses, those who successfully navigated the initial stages of the application process often encountered further frustration due to unclear communication and a

lack of constructive feedback from the administering body. These two compounding factors are probably why those farmers engaged in AES perceived application clarity to be worse than those not engaged and why advisers, who are often the individuals dealing with the AES application on behalf of the farmers, find application clarity to be worse than farmers. Improving this communication can build farmer's trust in AES (Sander et al., 2024) and over time can reshape farmers attitudes to be more positive towards AES (Riley., 2016). This combination of issues with the application process was emphasized in a recent consultation regarding future changes to Scottish agriculture where it was highlighted that current support is insufficient and there are gaps in delivery (Scottish Government, 2023).

5.4.4 Survey reliability

Farmers and advisers are notoriously time poor, so it was not feasible to gather a large group for a pilot study that could robustly check the Cronbach alpha of the Likert scales beforehand. Instead, the focus was placed on ensuring that the questions were appropriately framed and that all relevant areas of interest were covered, while keeping the survey duration under 10 minutes to avoid respondent fatigue. This was especially important as responses were being collected in busy public settings such as livestock auctions and agricultural shows, where a balance needed to be struck between the reliability of the Likert scales and the overall length of the survey.

The internal consistency of the Likert scales, as indicated by Cronbach's alpha scores, was acceptable given the exploratory nature of this study. In the context of social research aimed at identifying patterns rather than drawing definitive conclusions, moderate reliability levels are considered sufficient (Chan and Idris, 2017). The relatively lower alpha values observed for the social acceptance and information accessibility scales likely reflect the small number of items within these constructs—each consisting of only four statements. This aligns with established psychometric research showing that reliability tends to increase with the number of items, as the influence of individual item variance decreases in longer scales (DeVellis, 2012).

Most respondents were educated to university level, which is notably higher than the Scottish baseline of 33% (National Records of Scotland, 2024). This skew toward higher education levels is likely explained by the relatively high number of agricultural advisers in the sample, a professional group typically requiring formal qualifications. The majority of respondents were male, reflecting the broader gender distribution within Scottish agriculture, where 60% of farmers are male and 40% are female (Scottish Government. 2021).

A large proportion of respondents were located in East Central Scotland, likely influenced by in-person survey collection at a farming event held in Fife, which yielded particularly high response

rates. All in-person recruitment events were held within approximately a one-hour radius of Central Scotland, contributing to the limited representation from more distant regions such as the Northeast and Ayrshire. Crofters and new entrants also appear to be underrepresented in the sample, as indicated by the low number of responses from these groups.

In terms of farm characteristics, the respondent group appears broadly representative of the wider Scottish farming population. A significant number operated livestock-based enterprises in Less Favoured Areas, which reflects the dominant land use classification in Scotland (NFUS. 2025). Most respondents owned their farms, aligning with national patterns where 78% of land is owner-occupied and 22% is tenanted (Scottish Government. 2013). The median farm size reported was also close to the national average of 104 hectares (Scottish Government. 2023). Approximately 20% of Scottish farmers are currently enrolled in the Agri-Environment Climate Scheme (AECS) (NatureScot. 2023), and while participation data for other AES was unavailable, uptake among this sample appears to be above average.

Overall, the respondent population can be considered broadly representative of Scottish farmers and advisers, with a slight overrepresentation of individuals already engaged in AES and a location bias consistent with the use of both in-person and online survey methods.

5.5 Conclusion and Recommendations

This study has identified a range of farm demographic characteristics, attitudinal factors, and perceived barriers that influence participation in AES in Scotland. These findings provide valuable insight into why some farmers and land managers engage with AES while others do not, highlighting specific areas where policy and support mechanisms can be improved. In total, three key recommendations are proposed along with supporting evidence. Together, these recommendations offer a roadmap for delivering more targeted, flexible, and accessible schemes that better align with the realities of Scottish farming, ultimately supporting wider uptake and the successful delivery of environmental outcomes. A policy brief (appendix H) and infographic (appendix I) have also been created to summarise and translate this research into a more deliverable format.

Recommendation 1: Farmer led clusters

To address the issues associated with education, information, and social acceptance highlighted in Section 4.1, an enhanced fourth tier of the support package is recommended (Figure 21). This tier should be upgraded to provide a framework for the development of voluntary, farmer-led clusters across Scotland, building on the already established framework (GWCT, 2018). Direct financial and economic support for these farm clusters can be channelled through the 4th tier of the support package to enable their growth and continual development. To facilitate these groups, access to local and trusted advisory and technical support for the farm groups should be available. Further support from government agricultural extension services, NGOs, farmer organisations etc can then be integrated into this support package for the farmer clusters (Lastra-Bravo et al., 2015). This combination of public and private information and advice will increase farmers trust and knowledge of AES (Unay Gailhard et al., 2012) which will feed into greater participation (Tyllianakis et al., 2023).

Once established, farmer clusters can evolve to identify and co-produce the types of information and training most relevant to their specific farm systems and local landscapes, with help from external support. This collaborative approach helps to address key knowledge gaps, such as understanding which NbS are most effective in local conditions, how to navigate AES requirements, and where to access reliable guidance. It also fosters trust between farmers, advisers, policymakers, and environmental organisations by creating spaces for peer learning, shared decision-making, and open dialogue (Taylor and van Grieken, 2015). The farm clusters can then choose how they want the knowledge delivered, e.g., farm visits and walks, demonstration farms, meetings and show days, workshops etc. Empowering whole farming communities creates a snowball effect where neighbouring farmer's experiences and opinions influence others behaviour towards AES (Rodríguez-Entrena and Arriaza, 2013). These positive experiences affect farmers willingness to engage with AES

and reflects their growing confidence in being able to adapt to a system that suits them best (Lastra-Bravo et al 2015).

If effective, farmer clusters should act as catalysts for information exchange, enhancing environmental awareness of farmers (Lastra-Bravo et al., 2015) and making information accessible for all farmers (Ducos et al., 2009). Building these clusters with local farm communities at their heart will promote local social capital, which is a key factor in the adoption of AES (Burton and Schwarz, 2013; Burton and Paragahawewa, 2011; Lastra-Bravo et al., 2015). Social capital supports the development of trust, reciprocity, and shared expectations among farmers, creating a more enabling environment for knowledge exchange and collective action (Sutherland and Burton, 2011; Riley, 2016). It also helps reduce uncertainty, as farmers are more likely to engage with schemes that are endorsed and normalised within their peer networks (Marshall, 2009). Furthermore, these clusters can enhance the social acceptability of AES by facilitating interpersonal communication and collaboration between farmers (Unay Gailhard et al., 2012; Tyllianakis et al., 2023), which has been shown to be positively associated with higher levels of participation (Schaub et al., 2023).

Enabling farmers to engage and shape the objectives, methods and final outcome of AES is a powerful form of co-design (Hurley et al., 2022) that can bridge the priorities of farmers and AES policy implementers (Canessa et al., 2024). So, it is encouraging that the Scottish Government is engaging with the farmer-led climate groups as this will lead to better results by lowering uncertainties, and delivering policy oversights and knowledge asymmetries (Blokamp, 2018). To ensure the long term effectiveness of the new Scottish AES it is crucial that the farm clusters are involved in its design (Whittingham, 2011). If done correctly this process may further enhance AES participation by promoting engagement of the farmers who have traditionally been 'harder to reach' (Hurley et al., 2022) whilst also building buy in and trust in the final AES design (Mettepenningen et al., 2013; Canessa et al., 2024). Enhancing the social capital and acceptability of AES by having farmers working together at a landscape scale will also be important for Scotland's regional land use partnerships. Collaboration across neighbouring farms can help deliver landscape wide improvement to nature and climate (Tyllianakis and Martin-Ortega, 2021; Westerink et al., 2017) and facilitate AES participation (Schaub et al., 2023). Furthermore, it may open future avenues for blended finance options (Rodgers and Kendall, 2023). But this process needs to be done correctly and transparently to avoid farmers being unsure about collaboration which can negatively affect AES participation (Villamayor-Tomas et al., 2019, 2021).

These farm clusters will intersect with some of the key themes that the new agricultural knowledge and innovation system (AKIS) is trying to establish (ClimateXChange. 2023). Creating farm clusters at appropriate landscape or catchments levels will help intersect national policies with regional demands. Furthermore, having farmers lead and co-design the clusters will support peer to peer learning and farmer collaboration. Finally, the farmers can identify the key opportunities and upskilling they need and then can select from the options presented in AKIS such as monitor and demonstration farms, ambassador farmers field schools, mentoring etc. These farm clusters share similarities with the Environmental Clusters proposed by AKIS (ClimateXChange, 2023); however, a key distinction is that farm clusters follow a bottom-up, farmer-led approach. This empowers farmers to make decisions and helps build trust in AES and wider governmental institutions.

Recommendation 2: Removing risk and inflexibility from AES

Risk, and especially financial risk, needs to be removed from AES. The proposed agriculture support package from the Scottish Government offers a good starting point. Having a Tier 1 base level direct payment tied to sustainable farming behaviours, restoring the environment, animal health and welfare and safeguarding fair work ensures minimum standards and cross compliance (Figure 21). The reduced risk then comes from the tier 2 enhanced level direct payments that are universally accessible and deliver outcomes for nature and climate. Furthermore, tier 2 addresses flexibility by being outcome based which means that farmers can select from several different packages each with multiple measures to suit their needs whilst still achieving the goals set. Tier 3 will be competitive and non-universal, based around elective payments for specific habitats or species. At least half of all the funding for this new agricultural support package will be directed towards actions with outcomes which support climate mitigation, adaptation, and biodiversity gain (Scottish Government, 2024). However, the distribution of funding between these 4 tiers suggests that 70% of the total funding for agriculture will be allocated to the least ambitious tiers 1 and 2 with 70% of that going towards the universal entry-level payments (Scottish Government, 2025). This distribution of funding may not be enough to secure the levels of AES participation required for futureproofing food production against the impacts of the climate and nature crises (NFFN, 2025; Scottish Wildlife Trust, 2017). However, others have suggested that ensuring active farming alongside business support is the critical issue which supports the current funding split (NFUS, 2018; Royal Society of Edinburgh, 2024). It is recommended that the data and thought process behind the funding distribution of the four tiers should be made publicly available so that independent assessments can be made about how much funding is needed within each tier to meet Scotland's nature and climate targets whilst also securing financial stability for farmers and sustainable food production.

As a side note, information is still lacking on variable contract lengths (NFUS, 2018) and regionalised adaptation of the new AES which are important considerations (Zindler et al., 2024). It is presumed the latter will be delivered through regional land use partnerships (Scottish Government, 2022b).

Recommendation 3: Improving application clarity

Creating AES that are not overly complex with a simple application process and straightforward guidance has been identified as essential for the future of the Scottish agricultural support system (NatureScot, 2022; NFUS, 2018). Furthermore, the application process, communication and appeals process needs to be transparent, fair and consistent which will give confidence to farmers who wish to apply (Scottish Land & Estates, 2022). It is suggested that a system similar to the English Environmental Land Management (ELM) scheme be followed. The ELM application process aims to make the experience as smooth and straightforward as possible with an online portal, accessible and easy to understand guidance, and automation which creates an average processing time of 15 days (DEFRA, 2024a). This has resulted in higher participation rates across all the AES offered.

Additionally, results from farmers who were surveyed about their experience of the ELM's sustainable farming incentive (SFI) highlighted that 75% of research participants found it 'very easy' or 'quite easy' to apply and 81% of farmers that took part in research rated the existing SFI offer positively (DEFRA, 2024b). Creating an application process like this in Scotland will help to establish a streamlined and lean bureaucratic system where communication is transparent and timely which will break down barriers to participation (Cusworth & Dodsworth, 2021) so that farmers can hit the ground running when the new agricultural support package starts in 2027.

It is worth noting, however, that whilst the agricultural reform board and their reform route map has provided clear signposting to farmers and advisers on when the new AES will come into effect, farmers are still lacking sufficient detail on the payment system and the practicalities of implementation (Royal Society of Edinburgh, 2024)

6. Synthesis and key findings

6.1 Introduction to General Discussion

The overarching aim of this thesis was to assess the ecological and social dimensions of NbS within agricultural landscapes, with a specific focus on RSuDS and beaver-influenced systems. This aim was pursued through four interlinked objectives. The first objective was to assess how beaver activity modifies natural capital assets and ecosystem service provision relative to baseline conditions. This was addressed in Chapter 2 through expert elicitation and integration with national natural capital accounting, revealing that beavers contribute substantially to the enhancement of woodland ecosystem services at a national scale. The second objective was to quantify the range of ecosystem services and disservices provided by RSuDS, with particular attention to multifunctionality and trade-offs. Chapter 3 met this objective using the RAWES framework and spatial analysis, demonstrating how RSuDS deliver primarily regulating services, and identifying spatial and design factors that influence service delivery and associated trade-offs. The third objective focused on investigating greenhouse gas emissions from pond ecosystems to identify key drivers and contextualise these emissions as disservices. This was explored in Chapter 4 through field-based measurement and statistical modelling, which confirmed that agricultural ponds are net emitters of methane and carbon dioxide and provided insights into the environmental conditions driving these emissions. The fourth and final objective was to characterise the sociodemographic, perceptual, and structural barriers influencing the uptake of NbS on farms, and to identify opportunities to enhance adoption. Chapter 5 addressed this through a national survey, uncovering concerns about financial risk, scheme complexity, and lack of accessible guidance, and offered policy-relevant recommendations to improve the design and delivery of agri-environmental schemes. Collectively, these findings deliver on the overarching aim by providing a holistic assessment of both the ecological performance and social dimensions of NbS in agricultural landscapes, offering critical insights for more effective and inclusive nature-based policy and practice.

6.2 Synthesis of Main Findings

Several consistent themes emerged that together offer a deeper, systems-level understanding of NbS in agricultural landscapes. These themes reveal important insights about ecological processes, disservices, stakeholder considerations, and the socio-environmental contexts that shape the success of NbS (Figure 31).

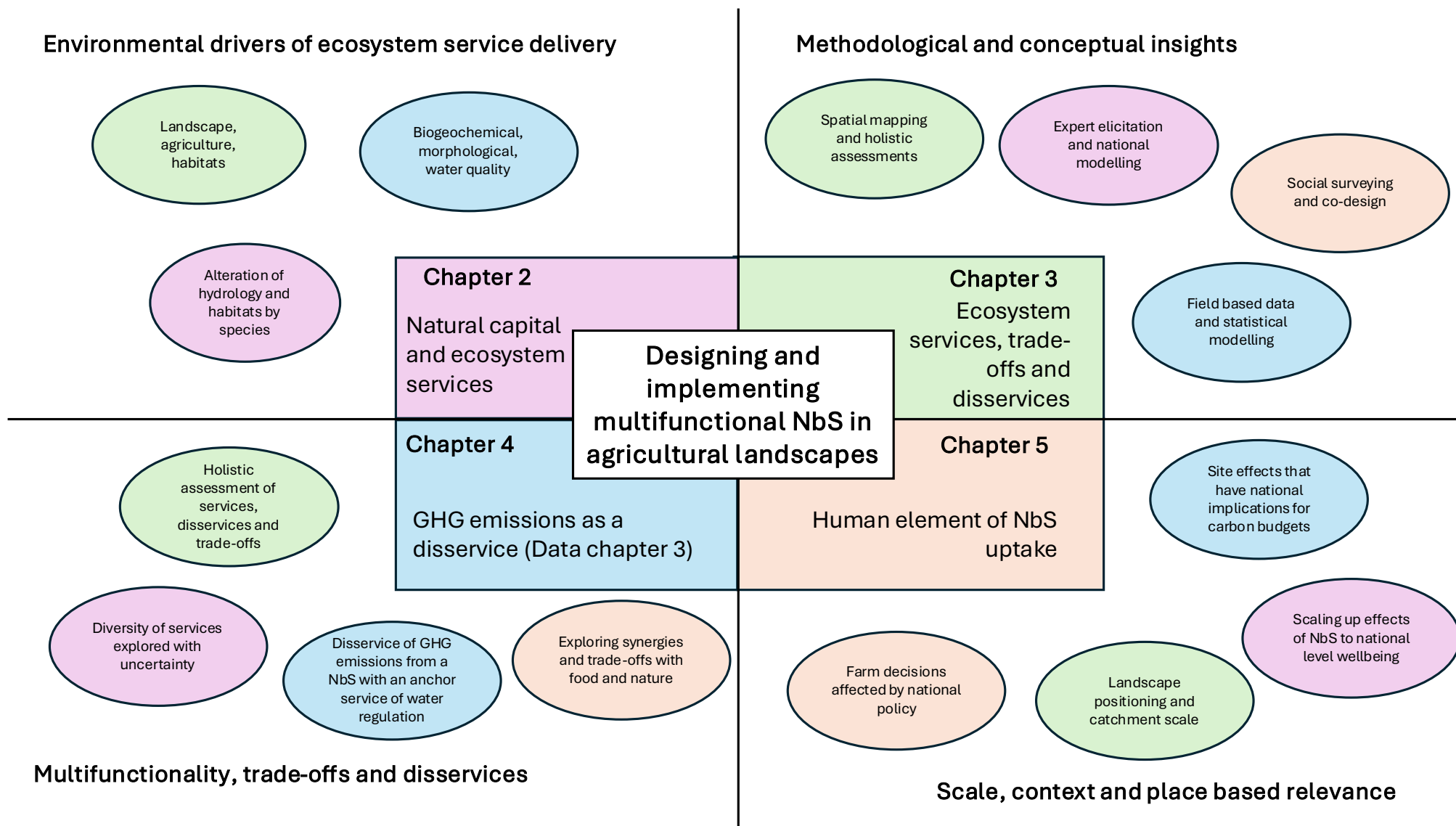


Figure 31: White box in the middle is the guiding aim for this thesis. Multicoloured boxes are the chapters that were undertaken. Colour of each chapter relates to the contributions to each of the themes identified

6.2.1 Environmental Drivers of Ecosystem Service Delivery

A consistent theme across the ecological components of this thesis is the importance of biophysical context in determining how NbS function. Environmental variables were found to exert strong influence over ecosystem service delivery. This reinforces a growing body of literature which argues that NbS are highly context-dependent and cannot be treated as universally effective interventions (Seddon et al., 2020; IUCN, 2020; Smith et al., 2023). Clear patterns emerged showing how ecological processes interact with local environmental conditions to mediate the performance of NbS.

For example, alterations to hydrological regimes and woodland structure were found to influence the capacity of landscapes to regulate water flow and support biodiversity (Chapter 2). These findings echo existing work showing that changes in flow dynamics can significantly shape the regulation of downstream services (Lane, 2017; Puttock et al., 2017). Similarly, the provision of water quality and flow-related services by RSuDS was found to be contingent on landscape positioning, feature design, and exposure to agricultural intensity (Chapter 2). These insights align with spatially explicit studies suggesting that service delivery from NbS is optimised when interventions are located strategically e.g. upstream of vulnerable receptors, near populated areas, or in zones of high runoff or nutrient export (Barber et al., 2021; Environment Agency, 2021). The work on pond GHG emissions (Chapter 3) highlighted the sensitivity of disservice generation to underlying biogeochemical drivers. This finding complements wider research showing that the net climate effect of wetland-based NbS varies dramatically with local water chemistry and sediment conditions (Holgerson and Raymond, 2016; Peacock et al., 2021).

Taken together, these findings contribute to a broader recognition in the NbS literature that ecological drivers are not just background variables, they are key to determining success, failure, or unintended consequences (Raymond et al., 2017; Seddon et al., 2021). The assumption that NbS will perform consistently across landscapes is increasingly being challenged and they must be tailored to the biophysical realities of place if they are to deliver multifunctional outcomes (IUCN, 2020; UNEP, 2022).

6.2.2 Multifunctionality, Trade-offs, and Disservices

A central theme emerging across this thesis is the inherent multifunctionality of NbS in agricultural landscapes, and the complexity of balancing this multifunctionality with trade-offs and disservices. NbS are widely recognised for their potential to deliver multiple ecosystem services simultaneously (Seddon et al., 2020; Nesshöver et al., 2017). This thesis supports that view, showing that well-

placed and well-designed interventions can generate co-benefits across water regulation, biodiversity support, and climate adaptation. However, it also reinforces that these benefits are neither uniform nor guaranteed, their magnitude, distribution, and perceived value are shaped by local ecological conditions and social priorities (Cohen-Shacham et al., 2019; Pascual et al., 2017).

Across the ecological studies, NbS features such as wetlands, ponds, and species-led interventions consistently delivered regulating services. However, provisioning and cultural services were often limited or under-recognised, especially by land managers, whose perceptions of value were more strongly linked to tangible outcomes like crop yields or financial return (Chapter 2). This mismatch between ecological function and stakeholder perception underscores the need for explicit management of trade-offs, rather than assuming alignment between service supply and societal benefit. This thesis also highlights the tension between ecosystem services and disservices, an area that remains underrepresented in NbS assessments (Zhang et al., 2007; Seddon et al., 2021). For instance, while constructed wetlands and ponds improved water quality and supported biodiversity, they also acted as net sources of greenhouse gases complicating assumptions about their climate mitigation potential (Chapter 3). This mirrors findings from global studies indicating that small water bodies can emit disproportionate levels of GHGs due to their high productivity and shallow depth (Holgerson & Raymond, 2016; Peacock et al., 2021).

Collectively, these findings support an emerging scientific consensus that NbS are not inherently “win-win” solutions, but rather complex interventions with differentiated outcomes (Raymond et al., 2017). Optimising multifunctionality while managing trade-offs requires a deep understanding of both ecological processes and social valuations. This thesis contributes to that understanding by demonstrating how multifunctionality emerges depending on site characteristics, management design, and stakeholder framing. In doing so, it reinforces calls to move beyond single-objective metrics (e.g. carbon sequestration) and instead evaluate NbS through integrated, context-sensitive frameworks (Everard et al., 2024).

6.2.3 Methodological and Conceptual Insights

A strong methodological theme running through this thesis is the integration of empirical ecological measurement with modelling and stakeholder-based approaches, reflecting the interdisciplinary nature of NbS research. The combination of field-based data, spatial analysis, structured expert input, and social survey methods allowed for a more complete understanding of how ecosystem processes, human values, and governance structures interact in agricultural landscapes. This triangulation of methods provided both site-level depth and system-level generality, enabling

ecological outcomes to be meaningfully situated within broader policy and land management contexts.

Such integration echoes calls for transdisciplinary approaches to sustainability science, particularly in the assessment of NbS where ecological effectiveness must be linked to social feasibility and institutional relevance (Lang et al., 2012; Raymond et al., 2017; Seddon et al., 2020). Isolated empirical measurements, while valuable, risk becoming decoupled from implementation pathways if not interpreted through policy-relevant frameworks. Conversely, governance recommendations without ecological grounding may result in interventions that are misaligned with landscape dynamics or poorly targeted.

The diversity of methodological approaches across the thesis enabled insights to emerge across ecological and social domains. This range of viewpoints made it possible to detect emergent properties of NbS, such as the spatial variability of ecosystem service delivery, or the disconnect between ecological performance and perceived value among stakeholders. It also revealed how different data types can complement rather than compete, when analysed in an integrated framework. At the conceptual level, this methodological integration supports a systems-based understanding of NbS as embedded socio-ecological interventions. It reinforces the growing consensus that environmental challenges cannot be addressed through disciplinary silos, and that effective NbS design and evaluation must operate across boundaries (Nesshöver et al., 2017; Chausson et al., 2020).

6.2.4 Scale, Context, and Place-Based Relevance

A consistent theme across this thesis is that the success and relevance of NbS are profoundly shaped by scale and context. NbS are not universally applicable interventions; rather, their performance, trade-offs, and societal value depend on how well they align with local environmental conditions, spatial dynamics, and institutional settings. This thesis demonstrates that ecosystem processes and stakeholder responses unfold differently at field, catchment, regional, and national levels, highlighting the importance of designing NbS with explicit attention to place-based specificity and cross-scale interactions.

At the ecological level, the scale of observation influenced which drivers of ecosystem service delivery were detectable and actionable. Biogeochemical processes governing greenhouse gas fluxes, for instance, operated at the microhabitat scale but had implications for national-scale emissions profiles (Chapter 3). Meanwhile, the influence of landscape positioning, feature size, and

proximity to receptors emerged as critical in shaping the multifunctionality of sustainable drainage features (Chapter 2). These findings align with broader research showing that NbS effectiveness is highly spatially variable, and that fine-scale heterogeneity must be accounted for in both modelling and planning (Kabisch et al., 2016; Chausson et al., 2020).

The social dimensions of scale were equally evident. Farmer engagement with NbS was shaped by localised perceptions of risk, land-use pressures, and access to advisory support, while broader institutional and governance structures determined the enabling conditions for uptake (Chapter 5). This reinforces the need to consider institutional scale alongside ecological scale in NbS design. As noted in recent work, misalignment between the scale of ecological processes and the scale of governance remains a key barrier to NbS effectiveness (Seddon et al., 2020).

The thesis illustrates that scaling up NbS, from site-level interventions to landscape or national strategies, cannot be achieved through replication alone. Outcomes are not simply additive; they are emergent and shaped by interactions between place-specific variables. The national-scale natural capital assessment of species-led restoration (Chapter 2) exemplifies how cumulative effects become visible at broader scales, but only if spatial and ecological variability are adequately captured. This supports growing recognition that scaling NbS requires nested approaches that balance local relevance with strategic coherence (IUCN, 2020; UNEP, 2022).

6.3. Research implications

6.3.1 Implications for scientific knowledge and understanding

This thesis advances the scientific understanding of NbS by extending their application into agricultural landscapes, an area historically under-represented in NbS research (Dunlop et al., 2024; Simelton et al., 2021). It directly responds to calls for empirical research that delineates the conditions under which NbS deliver their intended outcomes (Seddon et al., 2020). While NbS have gained global prominence as cost-effective, multifunctional interventions to address climate change, biodiversity loss, and sustainability challenges (IPBES, 2019; Cohen-Shacham et al., 2019), empirical evidence from production landscapes has remained limited. This thesis addresses that gap by demonstrating, through real-world case studies, how farm-scale NbS can simultaneously deliver multiple ecosystem services under variable environmental and management conditions.

The research challenges the often-assumed trade-off between intensive agriculture and ecosystem health, showing instead that strategic integration of NbS into farming systems can reduce tensions between food production, climate adaptation, and conservation. In doing so, it responds directly to

the lack of integrated evaluations that assess both ecological benefits and potential costs or trade-offs of NbS (Chausson et al., 2020). By empirically mapping and quantifying a broad suite of services and disservices, this work moves beyond narrowly focused assessments and provides a more holistic, system-wide evaluation.

This thesis explicitly acknowledges and analyses trade-offs. Few previous studies have systematically quantified the net delivery of services versus disservices over time or considered unintended outcomes such as reduced crop yields or altered hydrological processes (Seddon et al., 2020; Roe et al., 2021). This research helps fill that gap by providing robust, comparative evidence of agroecosystem multifunctionality in practice. It supports the premise that nature-based farm management can generate synergistic benefits, while also offering empirical insight into the tensions that can arise between competing objectives (Miralles-Wilhelm, 2023). In doing so, it reinforces and refines the multifunctionality paradigm by identifying where synergies are possible and where tensions persist, cautioning against simplistic assumptions that NbS are inherently win-win (IPBES, 2019). While this study does not encompass all forms of NbS, it provides a transferable approach that can be applied to other systems, offering a useful template for future assessments of multifunctionality and trade-offs.

Finally, this work contributes to redressing the imbalance in the evidence base, which has typically prioritised biodiversity and climate outcomes while overlooking broader societal benefits (Simelton et al., 2021). Through integrated assessment of NbS impacts on water quality, productivity, land management, and rural wellbeing, this thesis helps fill key knowledge gaps around the socio-ecological performance of NbS in working landscapes. It provides a transferable framework for evaluating multifunctionality and trade-offs, offering a valuable template for future research and policy development in agricultural and other production contexts.

6.3.2 Implications for Environmental Management and Conservation Practice

This thesis has shown that NbS such as beaver-engineered wetlands and RSuDS can significantly enhance ecosystem functions in agricultural landscapes. When appropriately designed and implemented, these interventions contribute to a suite of regulating services outcomes that are central to conservation goals at both site and landscape scales. These findings reinforce the ecological rationale for integrating NbS into broader strategies for environmental management and habitat restoration (FAO, 2020), while also responding to the urgent need for context-sensitive planning and implementation capacity (Martin et al., 2025).

However, this research also highlights that the delivery of these benefits is neither guaranteed nor uniform across contexts, underscoring the concern that many NbS projects proceed without

adequate design expertise, monitoring frameworks, or long-term operational planning (Corgo et al., 2024). For example, evidence that certain NbS may function as net sources of greenhouse gases under specific environmental conditions demonstrates the need for adaptive management frameworks grounded in local knowledge. Rather than assuming positive outcomes, this thesis supports the view that effective NbS implementation must be dynamic and responsive, requiring systematic monitoring, performance evaluation, and iterative design. These elements are often missing in current practice due to a lack of standardised protocols and accessible guidance (Dunlop et al., 2024), gaps that this research addresses by offering concrete data, design insights, and management considerations.

The case studies in this thesis provide practical insights into how site-specific variables influence NbS performance. This reflects the call for spatial targeting and the incorporation of ecological thresholds and biogeochemical feedbacks into system design (Seddon et al., 2020). By identifying where and how these interventions perform best, the research offers guidance that can support more deliberate planning by land managers, conservation practitioners, and local authorities. It responds to the widespread lack of technical capacity and ecological design expertise among implementation actors, helping to bridge the knowledge-practice gap through empirically grounded evidence (European Environment Agency, 2023).

The research also addresses the growing recognition that NbS implementation involves not just ecological planning but also conflict mediation, particularly in cases of species-led interventions (Cusack et al., 2021). For example, ecosystem engineering by beavers demonstrates the dual potential of NbS to generate widespread ecological benefits and localised disservices. This reinforces the need for governance structures that can respond flexibly to emergent outcomes and stakeholder concerns, an issue often overlooked in conventional conservation planning (IUCN, 2020; Berg et al., 2024). Rather than viewing such interventions as passive or hands-off, this thesis argues for their integration into adaptive ecosystem management frameworks (Mansson et al., 2023).

6.3.3 Implications for Policy and Strategic Decision-Making

NbS are increasingly central to the design of post-Brexit environmental governance and land-use policy in the UK and beyond (Environment Agency, 2025). However, their integration into policy and decision-making frameworks remains fragmented and often hampered by institutional inertia and sectoral silos (Hölscher et al., 2023). This thesis directly addresses those limitations by providing context-specific, empirically grounded evidence that supports the operational mainstreaming of NbS within next-generation agri-environmental schemes and broader climate mitigation strategies.

By demonstrating that well-designed NbS can deliver a range of ecosystem services this research strengthens the empirical foundation for policy mechanisms that reward ecosystem service delivery. In doing so, it contributes to overcoming one of the key policy barriers identified in the literature: the limited and uneven evidence base on the effectiveness and efficiency of NbS relative to engineered alternatives (Seddon et al., 2020; Chausson et al., 2020). Conventional frameworks often struggle to account for the multi-functional and non-market benefits of NbS, resulting in their under-valuation in policy assessments (Wild et al., 2017; Pascual et al., 2017). This research responds by using multi-indicator assessments that highlight the range of ecosystem services delivered by NbS, including those that are difficult to monetise. These findings can inform the development of more integrated policy appraisal approaches that better reflect the diverse values associated with nature-based approaches.

At a strategic level, this thesis supports the integration of NbS into regional and national sustainability planning. Its spatially explicit findings offer tools for identifying priority areas for intervention, optimizing ecosystem service delivery, and minimizing conflicts with productive land use. These contributions reinforce and can strengthen the implementation of evolving policy frameworks, including Local Nature Recovery Strategies (LNRS), which provide spatially targeted habitat and NbS planning guidance (POST, 2021; Environment Agency, 2024); mandatory Biodiversity Net Gain (BNG) (MHCLG/Defra, 2024); and catchment-based approaches that coordinate NbS across hydrological and ecological boundaries (Environment Agency, 2024). They also support the broader policy transition toward multifunctional landuse planning, where food production, ecological resilience, and climate goals are addressed in a coordinated manner rather than through siloed initiatives.

6.3.4 Implications for Stakeholder Engagement and Environmental Stewardship

Effective stakeholder engagement is fundamental to the successful implementation and long-term sustainability of NbS in agricultural landscapes. Many NbS initiatives suffer from limited or superficial engagement, often confined to initial consultations or end-of-project wrap-up phases (Brill et al., 2022; Bogatinoska et al., 2022). This research directly addresses those limitations by demonstrating that the viability of NbS depends not only on ecological performance but also on how interventions are perceived, co-created, and maintained by those who manage and inhabit the land.

The findings show that while many farmers and land advisers are broadly supportive of environmental action, uptake of NbS is frequently constrained by procedural complexity, perceived financial risk, and lack of access to clear, context-appropriate guidance. These barriers mirror the

structural and relational constraints identified in the literature, where top-down governance, weak advisory infrastructure, and short-term project cycles have often undermined the development of lasting stewardship cultures (Holstead et al., 2017; Mugari et al., 2025).

This research demonstrates that co-designed approaches can enhance the legitimacy, relevance, and durability of NbS, directly addressing research gaps around sustained engagement and farmer-led innovation. By aligning ecological interventions with landholder priorities and local agronomic knowledge, participatory models foster ownership and reduce resistance to change (Liu and Martens, 2023; van de Gevel et al., 2020). They also strengthen trust, which is critical for navigating trade-offs and securing long-term stewardship commitments (Coeugnet et al., 2023; Bogatinoska et al., 2022). Evidence from this study supports emerging practice-based models—including co-design workshops, advisory cooperatives, and peer-to-peer learning platforms—that have been shown to accelerate the adoption and upscaling of environmental practices (Zhang et al., 2024). Crucially, these approaches enable sustained dialogue beyond the lifespan of individual projects, a feature identified as lacking in many NbS initiatives (Brill et al., 2022). The emphasis on technical knowledge, advisory capacity, and peer learning mirrors the successes of programmes such as the Agricultural Catchments Programme and the Demonstration Test Catchments initiative (Old et al., 2020), both of which illustrate that effective policy support must extend beyond financial incentives to include knowledge systems that promote co-learning and institutional trust, essential conditions for scaling NbS from pilot to mainstream practice.

Operationalising these insights requires investment in enabling infrastructure. The findings highlight that advisory capacity, simplified compliance processes, and embedded feedback mechanisms are central to effective stakeholder engagement. These priorities address the institutional constraints and capacity deficits outlined in the introduction, where insufficient practitioner engagement and under-resourced support systems have limited NbS scalability (Mugari et al., 2025; Martin et al., 2025). Overall, the evidence suggests that successful stewardship is more likely to arise from well-resourced, relationally grounded support structures than from top-down, compliance-driven schemes.

6. 4. Critical Evaluation of Thesis Strengths and Limitations

This thesis adopts a deliberately interdisciplinary and multi-method approach to evaluating NbS in agricultural landscapes. It integrates ecological modelling, biogeochemical measurement, spatial analysis, and stakeholder research to examine both the biophysical functioning and social dimensions of NbS. Together, these approaches provide a robust foundation for the synthesis and recommendations offered. At the same time, the thesis is not without its limitations.

6.4.1 Methodological Strengths

A key methodological strength of this thesis lies in its triangulation of approaches across ecological, biophysical, and social domains. By combining field-based ecosystem service appraisal, spatial analysis, causal modelling, and stakeholder survey data, the research constructs a robust and multidimensional evidence base. This integration allows for a more holistic assessment of NbS, capturing both their ecological functioning and the socio-institutional conditions that influence their uptake and performance. The use of diverse and complementary methods enabled both depth and breadth in the analysis of NbS. Quantitative ecological assessments were reinforced by advanced analytical tools, including spatial regression and structural equation modelling, which not only allowed for rigorous estimation of service delivery and disservices but also facilitated the identification of causal pathways and the interactions between environmental drivers. The inclusion of methods more aligned with social science approaches (e.g., expert elicitation, surveys) provided essential insight into the human dimensions of NbS implementation.

A further strength of the research was the collaboration with external partners, which enhanced both the methodological design and the applicability of the findings. The natural capital and beaver assessment was undertaken in partnership with NatureScot, enabling access to datasets, expert networks, and policy-relevant modelling frameworks that substantially increased the scope and credibility of the analysis (Chapter 2). Similarly, the national farmer and adviser survey was developed in collaboration with the Scottish Farm and Wildlife Advisory Group (ScotFWAG), whose established relationships within the agricultural community facilitated high response rates, improved the representativeness of the sample, and ensured that survey instruments addressed issues of direct relevance to practitioners (Chapter 5).

By capturing perceptions, motivations, and institutional barriers alongside biophysical outcomes, the thesis delivers a richer understanding of the contextual factors shaping the feasibility and effectiveness of ecological interventions in real-world settings. The capacity to cross-validate findings across semi-qualitative and quantitative domains improves confidence in their coherence and generality. Collectively, the methodological approach exemplifies a systems-oriented research

design capable of capturing complex interactions between ecosystems, land use, and governance, and demonstrates how integrating ecological measurement, stakeholder engagement, spatially explicit modelling, and strategic partnerships can yield comprehensive insights into the performance and potential of NbS.

6.4.2 Limitations and Constraints

A primary limitation of findings emerging from this thesis concerns the issue of generalisability . While the ecological case studies were designed to capture key environmental gradients and management contexts, they were largely based in Scotland and often focused on specific landscape types or seasonal periods. This may limit the extent to which the results can be extrapolated to other geographical regions, land-use systems, or climatic conditions.

The social research also encounters representational constraints. The stakeholder survey, while broad in geographic coverage, may underrepresent certain demographic groups, such as younger or more digitally excluded land managers, and may be biased towards respondents already engaged with agri-environmental schemes. Similarly, the natural capital valuation of beaver activity was based on structured expert elicitation. While this method offers a robust approach to synthesising expert knowledge, it remains inherently subjective and contingent on the framing and diversity of the expert panel, despite efforts to apply methodological safeguards.

Temporal scale presents another important constraint. Much of the ecological data, particularly on greenhouse gas fluxes and ecosystem service dynamics, were collected over a single field season. While this captures high-activity periods and offers valuable insight into seasonal processes, it inevitably limits the detection of interannual variability, lag effects, or long-term trends. This is particularly relevant for NbS, many of which deliver benefits or reveal trade-offs over extended timeframes. For example, the ecological and hydrological consequences of beaver-driven wetland development may evolve over decades. That said, these constraints are characteristic of PhD research, and the findings presented here offer a strong foundation for future longitudinal studies. Addressing these knowledge gaps will require larger, well-resourced, and multidisciplinary research teams capable of sustaining long-term monitoring and integrating diverse expertise across ecological, hydrological, and social domains.

Finally, the integration across chapters, while conceptually coherent, required synthesising data collected using different methods, metrics, and spatial-temporal scales. Ecosystem service assessments, for instance, employed varying valuation frameworks depending on context and data availability, making direct comparison and aggregation analytically challenging. Similarly, not all ecosystem services or disservices could be evaluated with equal rigour across sites or intervention

types. These challenges reflect the complexity of interdisciplinary environmental research, where methodological pluralism is often necessary to address diverse questions (Millar et al., 2008).

6.5. Future research

While this work addresses several pressing gaps in the current literature, it also reveals important avenues for further investigation that would strengthen the evidence base and practical implementation of NbS in agricultural landscapes. These recommended priorities (Table 14) are organised as a short- to long-term roadmap, reflecting both achievable near-term advances and more ambitious, longer-term goals needed to embed NbS within resilient socio-ecological systems.

Table 14: Recommended future research

Priority Theme	Short-term goals (≤5 years)	Long-term ambitions (>5–10 years)
1. Capturing Cultural Ecosystem Services (CES)	Develop and test participatory, mixed-method tools, such as participatory GIS (Brown & Fagerholm, 2015), community mapping, narrative elicitation (Niemiec et al., 2021), to spatially and qualitatively capture CES linked to place identity, heritage, and recreation (Chan et al., 2012; Tratalos et al., 2016). Pilot co-designed indicators with local communities to reflect non-market values in NbS evaluations (Pascual et al., 2017; Natural England, 2020).	Embed CES metrics into standard ecosystem service assessment frameworks used by governments and NGOs (TEEB, 2010; UK Government, 2022). Integrate CES into national natural capital accounts, enabling policy decisions that reflect cultural as well as biophysical benefits.
2. Longitudinal Monitoring of NbS	Initiate multi-season monitoring programs to assess short-term variability in ecological and social outcomes of NbS (Seddon et al., 2020). Develop low-cost continuous sensing and citizen science protocols to fill seasonal data gaps.	Establish long-term (>10 year) monitoring networks across diverse agricultural landscapes to detect persistence, resilience, and lag effects of NbS under climate variability and policy change (Perring et al., 2015).
3. Broadening Disservice Assessment	Extend field-based assessments to include a wider suite of potential disservices (e.g., disease vectors, invasive species, altered hydrology) alongside benefits (Zhang et al., 2007). Incorporate these into ecosystem service mapping tools.	Develop predictive models that integrate services and disservices for different NbS types under multiple scenarios, enabling precautionary, risk-adjusted planning at national and regional scales.

Priority Theme	Short-term goals (≤5 years)	Long-term ambitions (>5–10 years)
4. Coupling Ecological and Social Models	Build prototype coupled models linking land manager decision-making with biophysical service flows to test uptake thresholds and explore policy sensitivity (Ostrom, 2009; Lade et al., 2013). Use empirical data from case studies to parameterise behavioural components.	Operationalise coupled socio-ecological models in decision-support platforms for policymakers and practitioners, allowing dynamic simulation of NbS outcomes under real-world policy, market, and climate conditions (Meadows, 2008).
5. Evaluating Policy Mechanisms	Conduct real-time evaluations of new agri-environmental schemes, comparing performance across governance contexts (Reed et al., 2018). Use adaptive learning cycles to feed findings back into scheme design (Folke et al., 2005).	Institutionalise adaptive governance frameworks for NbS that incorporate continuous learning from comparative studies, ensuring policy remains flexible, equitable, and ecologically effective over decades (Chaffin et al., 2014).

Conclusion

This thesis has explored the role of nature-based solutions (NbS) in agricultural landscapes through a multidisciplinary investigation of their ecological functionality, trade-offs, and social dimensions. By combining ecosystem service assessment, biogeochemical measurement, and stakeholder analysis, it contributes to a more integrated understanding of how NbS can support sustainable land management in practice. The findings highlight both the promise and complexity of NbS, showing that they can play a vital role in reconciling food production with environmental sustainability, but only if they are designed and managed with a clear understanding of ecological processes, social dynamics, and institutional realities.

Collectively, the thesis advances the case for systems thinking in NbS research and practice. Rather than treating ecological and social elements in isolation, it demonstrates the value of interdisciplinary approaches that reflect the complexity of real-world agricultural landscapes. It contributes empirical evidence to ongoing policy debates in the UK and beyond and offers conceptual tools for aligning environmental interventions with practical and social feasibility.

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Appendices

Appendix A

Ecosystem area ('000 Hectares)		2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
G. WOODLAND, FOREST AND OTHER WOODED LAND	G1 Broadleaved deciduous woodland	235	238	242	244	248	251	252	254	257	294	298	305	309	311	312	314	318	320	352
	G1 Broadleaved deciduous woodland - potential beaver woodland	82	83	85	85	87	88	88	89	90	103	104	107	108	109	109	110	111	112	123
	G1 Broadleaved deciduous woodland - remaining woodland	153	155	157	159	161	163	164	165	167	191	194	198	201	202	203	204	207	208	229
	G4 Mixed deciduous and coniferous woodland	42	42	42	42	42	42	42	42	42	42	42	42	42	42	42	42	42	42	42
	G4 Mixed deciduous and coniferous woodland - potential beaver woodland	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15
	G4 Mixed deciduous and coniferous woodland - remaining woodland	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28

Appendix B

What country do you primarily study or work in?

- ☐ Albania
- ☐ Andorra
- ☐ Austria
- ☐ Belarus
- ☐ Belgium
- ☐ Bosnia and Herzegovina
- ☐ Bulgaria
- ☐ Croatia
- ☐ Cyprus
- ☐ Czech Republic
- ☐ Denmark
- ☐ Estonia
- ☐ Finland
- ☐ France
- ☐ Germany
- ☐ Greece
- ☐ Hungary
- ☐ Iceland

- ☐ Ireland
- ☐ Italy
- ☐ Kosovo
- ☐ Latvia
- ☐ Liechtenstein
- ☐ Lithuania
- ☐ Luxembourg
- ☐ Malta
- ☐ Moldova
- ☐ Monaco
- ☐ Montenegro
- ☐ Netherlands
- ☐ North Macedonia
- ☐ Norway
- ☐ Poland
- ☐ Portugal
- ☐ Romania
- ☐ Russia
- ☐ San Marino
- ☐ Serbia
- ☐ Slovakia
- ☐ Slovenia
- ☐ Spain
- ☐ Sweden
- ☐ Switzerland
- ☐ Turkey
- ☐ Ukraine
- ☐ United Kingdom
- ☐ Vatican City

Please select the main research fields you work in (can select multiple)

- ☐ Ecology
- ☐ Hydrology
- ☐ Wildlife Biology

- ☐ Zoology
- ☐ Conservation Biology
- ☐ Environmental Science
- ☐ Behavioural Ecology
- ☐ Evolutionary Biology
- ☐ Landscape Ecology
- ☐ Geomorphology
- ☐ Sociology
- ☐ Anthropology
- ☐ Economics
- ☐ Geography
- ☐ Political Science
- ☐ Psychology
- ☐ Other

If other, please specify

What is your full name? (we require this for co-authorship and to know who has completed the survey)

What institution or organisation do you work in?

Would you like to be a co-author on any papers that are published from this work?

☐ Yes

Overview

Hello and thank you for taking the time to complete this survey :)

Here is a very brief overview of what to expect:

1. Training = on how to give answers in the correct format for the expert elicitation process we have chosen
2. Calibration questions = you'll be asked questions we know the answers to but you should not, to test how much you know about beavers. These are meant to be hard so don't worry!
3. Questions of interest = you'll be asked to judge how beavers effect the ecosystem services from woodlands and inland surface waters

Training - How to answer questions

Here you will learn how to give answers based on probability quantiles

The expert elicitation process you will be using is called Cookes classical method and uses probability quantiles (Figure 1). For each question you will give numbers for three answers:

- 5% quantile = you believe there is a 5% chance that the number would be lower than the 5% quantile. Think of this as the lowest number you think it could be.
- 95% quantile = you believe there is a 5% chance that the number would be greater than the 95% quantile. Think of this as the highest number you think it could be.
- 50% quantile = you believe this number to be the average or mean. Think of this as what number it's most likely to be.

In summary, the 3 answers are your low guess, high guess and best guess.

Asking questions in this format allows us to quantify the uncertainty of answers. Instinctively, you might want to answer the 50% quantile first but please answer the 5% and 95% first.

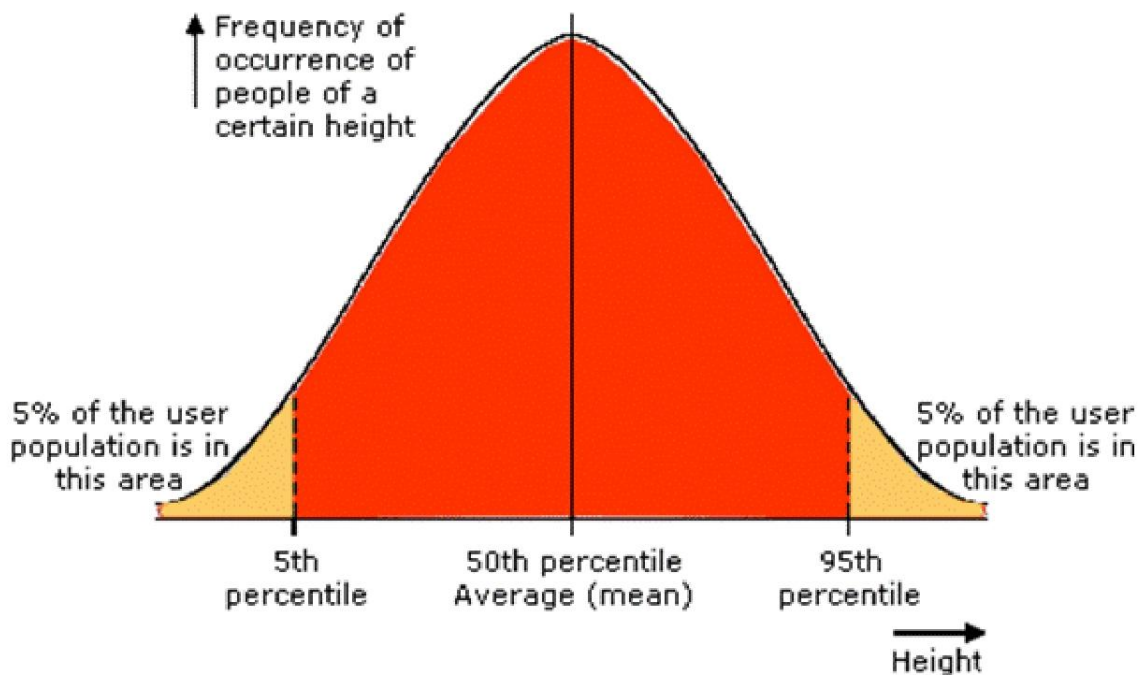


Figure 1: Probability distribution with quantiles

Between the 5% and 50% quantiles, you believe there is a 45% chance that the number would be between the 5% and 50% quantile. Between the 50% and 95% quantiles, you believe there is a 45% chance that the number would be between the 50% and 95% quantile.

Remember that values outside your 90% interval are still possible but you would be surprised to see them. We are not asking for a full range of possible values. By definition, a value outside your 90% interval might be observed one out of ten times.

Training - Calibration questions

Here you will learn about calibration questions and a few example questions are provided at the end.

Initially, you will be asked 10 questions that we already know the answer to, but you should not.

These are meant to be challenging! Please do not search for the answers but provide an informed judgement. These are called calibration questions because we are assessing your performance as an expert. Based on your response to the calibration questions you will be scored on your statistical accuracy and informativeness (Figure 2). Less informative but more accurate answers are more important and will give you a better score.

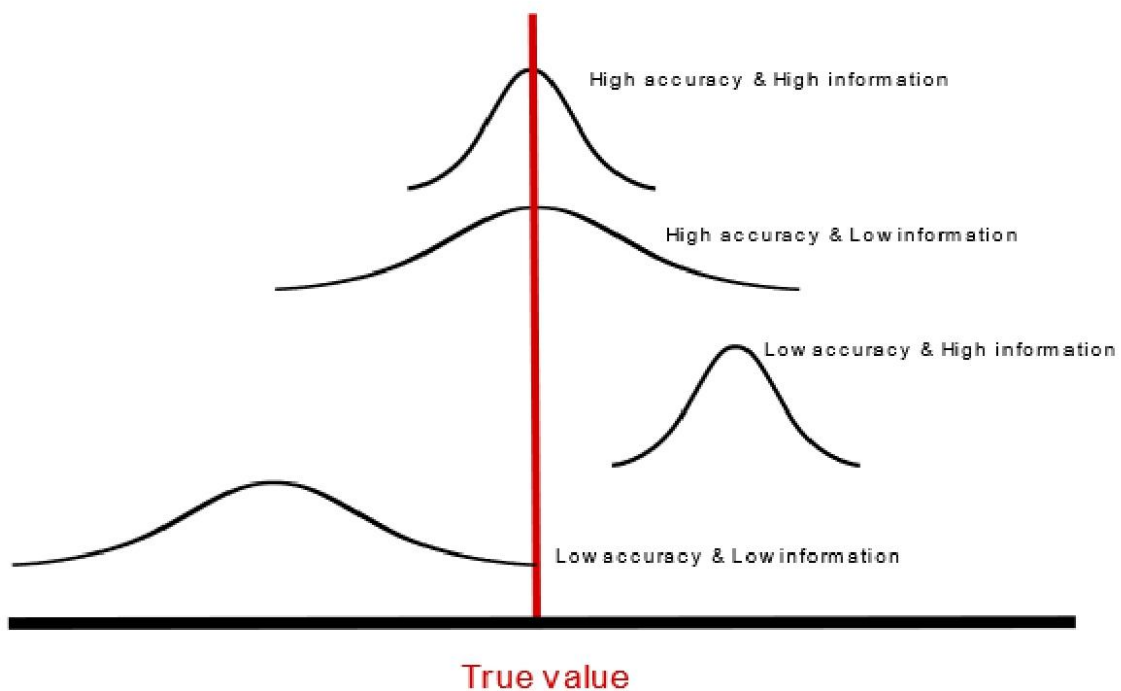


Figure 2: Different configurations of statistical accuracy and informativeness based off an experts judgement.

After each calibration question, you should explain your thinking, including the data, assumptions, or scenarios that influenced your uncertainty. Providing qualitative explanations helps understand study results, compare expert assessments, and uncover differences in interpretation.

It is important to answer these calibration questions to the best of your ability. This is because your responses to the actual questions will be weighted based on how you performed during the calibration questions. A higher performance = a higher weighting and so the final numbers agreed upon (decision maker) will be pulled more towards those who performed better (Figure 3).

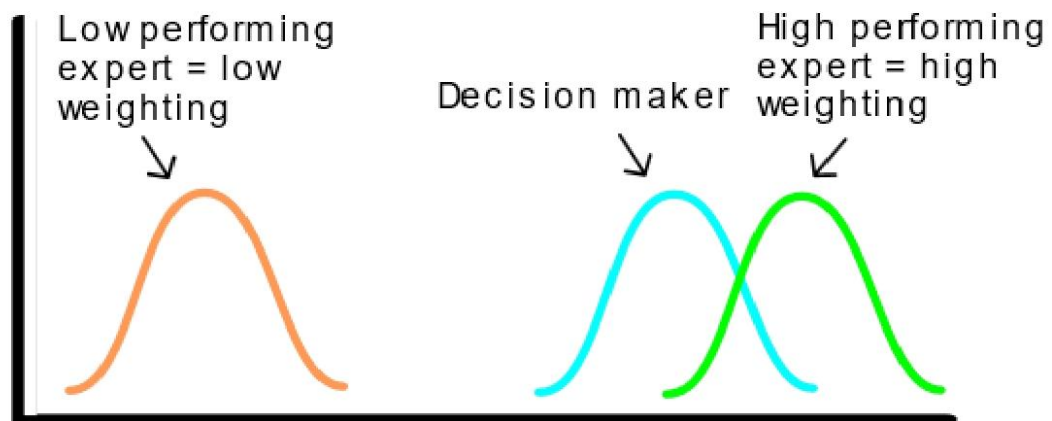


Figure 3: How the weighting of an expert affects the decision maker

Example: According to Cromsigt et al., (2018) European bison reduce the encroachment of woods through herbivory. What percentage of their diet throughout the year is composed of woody plants? 5% Quantile

Example: What percentage of their diet throughout the year is composed of woody plants? 95% Quantile

Example: What percentage of their diet throughout the year is composed of woody plants? 50% Quantile

Please explain your thinking behind these 3 answers

☐ No

Answer to calibration question example

According to Cromsigt et al., (2018) European bison reduce the encroachment of woods through herbivory. What percentage of their diet throughout the year is composed of woody plants?

Answer = 20%



Training - Questions of interest

 No page description set



You will now learn how to give answers to the actual questions, also called the questions of interest. We want to know what effect beavers have on the ecosystem services delivered by inland surface waters and woodland. You will be asked 15 questions and must give a number from -1 to 1 for the 5%, 50% and 95% quantiles.

- Values closer to -1 mean you believe beavers will have a very large negative effect on that ecosystem service.
- Values closer to 1 mean you believe beavers will have a very large positive effect on that ecosystem service.
- Values closer to 0 mean you believe that beavers will have minimal or no positive/negative effect on that ecosystem service,

The 15 questions are split into two parts. The first part asks you what effect beavers will have on ability of inland surface waters (lakes, rivers, ponds) to deliver ecosystem services. The second part asks the same questions but for woodland. Again, we ask you after each question to explain your thinking, including the data, assumptions, or scenarios that influenced their uncertainty. Some questions cover broad topics, if you know a lot about some part of that topic but not a lot of the rest of it then please say so.



12. Example: What effect will bison have on pollination and seed dispersal in woodlands?
5% Quantile

Please select an option ▾



13. Example: What effect will bison have on pollination and seed dispersal in woodlands?
95% Quantile

Please select an option ▾



14. Example: What effect will bison have on pollination and seed dispersal in woodlands?
50% Quantile

Please select an option ▾



15. Please explain your thinking behind these 3 answers



16. Example: What impact will otters have on the physical and experiential interactions (activities like wildlife observation and hiking promotes well-being, supports local economies, and fosters appreciation for nature) delivered by inland surface waters?
5% quantile

Please select an option ▾



17. Example: What impact will otters have on the physical and experiential interactions delivered by inland surface waters? 95% quantile

Please select an option ▾



18. Example: What impact will otters have on the physical and experiential interactions delivered by inland surface waters? 50% quantile

Please select an option ▾



19. Please explain your thinking behind these 3 answers

Answers to questions of interest examples

What effect will bison have on pollination and seed dispersal in woodlands?

We don't know the exact answer to this question but we can use our knowledge to make a reasonable guess.

Example: I know that large herbivores enhance seed dispersal, transporting seeds in their dung and fur. So straight away I believe they will have a positive impact on seed dispersal. I then recall a paper from Jaroszewicz et al., (2008) where the intensity of seed dispersal was 3.5 times greater in bison impacted areas, indicating this positive impact is significant. As for pollination, I know that herbivory from large herbivores increases plant and insect diversity. I then remember a paper from Bruninga-Socolar et al (2022) where they found bison grazing supported wild bee populations, which act as pollinators. So, I now know that bison have a positive impact on both pollination and seed dispersal so my 5%, 50% and 95% quantiles would be closer to 1 (significant positive impact) than 0 (minimal or no impact).

When asked to explain your reasoning for the actual question answers you can be more concise and use bullet points, this is just an example of the thought process.

This is an example of an ecosystem service where there is enough scientific evidence to have a degree of certainty in the the quantiles selected, so your informativeness might be high.

What effect will otters have on the physical and experiential interactions delivered by inland surface waters?

We don't know the exact answer to this question but we can use our knowledge to make a reasonable guess.

Example: I don't know of any specific information that mentions the effect otters have on physical and experiential interactions. But I know they are a cultural symbol of environmental importance relating to water. Furthermore, I can think of how symbolic creatures such as eagles and ospreys have positive effects on ecosystem services through ecotourism. I think that otters will have a similar effect but I am not entirely sure. So, I think that otters have a positive impact but I am not certain so my 5%, 50% and 95% quantiles would be closer to 0 (minimal or no impact) than 1 (significant positive impact).

This is an example of an ecosystem service where there is a lack of scientific evidence, and so selected quantiles may be less informative to capture the uncertainty. This is common in cultural ecosystem services because they are more difficult to quantify and are unrepresented in the scientific literature.

Appendix C

Calibration question 1

- Neumayer et al., (2020) modelled the impact of 12 beaver dams in two catchments over 8 storm events on peak flows. What percentage reduction of the peak flow did they find for flood events with a return period of less than two years?



20. What percentage reduction did beavers have on peak flow? 5% quantile



21. What percentage reduction did beavers have on peak flow? 95% quantile



22. What percentage reduction did beavers have on peak flow? 50% quantile



23. Please explain your thinking behind these 3 answers

👁 **Calibration question 2**

- Puttock et al., (2017) investigated the sediment retention capacity of 13 beaver dams on a first-order tributary draining from agricultural land. The average suspended sediment entering the dams was 112 ± 72 mg/L. What was the average suspended sediment average leaving the dams? (mg/L)



24. What was the average suspended sediment average leaving the dams? (mg/L) 5% quantile



25. What was the average suspended sediment average leaving the dams? (mg/L) 95% quantile



26. What was the average suspended sediment average leaving the dams? (mg/L) 50% quantile



27. Please explain your thinking behind these 3 answers

👁 **Calibration question 3**

Johnston (2014) found that carbon density to a depth of 60 cm was significantly greater in beaver meadows than in adjacent forest soils that had not been

- 👁 impounded. Adjacent forest soils that had not been impounded had an average carbon density of $8.2 \pm 2.9 \text{ kg m}^{-2}$. What was the average carbon density within the beaver meadows (kg m^{-2})?



28. What was the average carbon density within the beaver meadows (kg m^{-2})? 5% quantile



29. What was the average carbon density within the beaver meadows (kg m^{-2})? 95% quantile



30. What was the average carbon density within the beaver meadows (kg m^{-2})? 50% quantile



31. Please explain your thinking behind these 3 answers



Calibration question 4

- Elmeros et al., (2003) monitored the reintroduction of beavers in Denmark. What % of beaver cut trees were found within 5m (15ft) of watercourses (brooks, streams and small rivers) ?



32. What % of beaver cut trees were found within 5m (15ft) of water? 5% quantile



33. What % of beaver cut trees were found within 5m (15ft) of water? 95% quantile



34. What % of beaver cut trees were found within 5m (15ft) of water? 50% quantile



35. Please explain your thinking behind these 3 answers

👁 **Calibration question 5**

Fairfax et al., (2024) investigated if beaver-modified river areas are more resistant to wildfires. They found that riverbanks with beaver dams suffer less severe burns compared to those without,

- 👁 indicating that beaver-influenced landscapes are more resilient to wildfires. 60% of riverscapes without beaver dams were classified as fire refugia, what percentage of beaver-dammed riverscapes were classified as fire refugia?



36. What percentage of beaver-dammed riverscapes were classified as fire refugia? 5% quantile



37. What percentage of beaver-dammed riverscapes were classified as fire refugia? 95% quantile



38. What percentage of beaver-dammed riverscapes were classified as fire refugia? 50% quantile



39. Please explain your thinking behind these 3 answers

⦿ **Calibration question 7**

- Weber et al., (2017) monitored water temperature throughout 34 km of streams for an eight-year period between 2007 and 2014 In Oregon, USA. What reduction in the mean daily relative temperature (Celsius) did they observe in beaver-impacted streams compared to unimpacted streams?



44. What reduction in the mean daily relative temperature (Celsius) did they observe in beaver-impacted streams compared to unimpacted streams? 5% quantile



45. What reduction in the mean daily relative temperature (Celsius) did they observe in beaver-impacted streams compared to unimpacted streams? 95% quantile



46. What reduction in the mean daily relative temperature (Celsius) did they observe in beaver-impacted streams compared to unimpacted streams? 50% quantile



47. Please explain your thinking behind these 3 answers

☉ **Calibration question 8**

Ulicsni et al., (2020) interviewed 90 local people in central Europe about the perceived effect that beavers have on ecosystem services and livelihoods. When answering questions about whether beavers have a harmful/ambivalent/beneficial effect on ecosystem services and livelihoods only 5% believed beavers were beneficial. What percentage of respondents regarded beavers as harmful?



44. What percentage of respondents regarded beavers as harmful? 5% quantile



45. What percentage of respondents regarded beavers as harmful? 95% quantile



46. What percentage of respondents regarded beavers as harmful? 50% quantile



47. Please explain your thinking behind these 3 answers

Calibration question 9

- A study of experts involved in the research and/or management of beaver populations in mainland Europe found that most conflicts appeared to be from beavers foraging on crops (Campbell et al., 2007). What percentage of respondents reported that this occurred in their region?



52. What percentage of respondents reported that this occurred in their region? 5% quantile



53. What percentage of respondents reported that this occurred in their region? 95% quantile



54. What percentage of respondents reported that this occurred in their region? 50% quantile



55. Please explain your thinking behind these 3 answers

Calibration question 10

- Auster et al., (2019) surveyed the UK population on attitudes towards legal protection and management responsibilities of beavers. 2759 submissions were received, what percentage of the respondents supported reintroduction?



56. What percentage of the respondents supported reintroduction? 5% quantile



57. What percentage of the respondents supported reintroduction? 95% quantile



58. What percentage of the respondents supported reintroduction? 50% quantile



59. Please explain your thinking behind these 3 answers

⑥ **Symbolic, sacred and/or religious interactions**

Refers to the profound cultural and spiritual connections people have with plants, animals, landscapes, and natural events. This includes emblematic species like the American eagle or the English rose, which hold deep cultural significance. Ecosystems also serve as spiritual spaces with sacred sites and pilgrimage routes revered by religious and indigenous groups. Certain plants and animals are considered sacred and are used in religious ceremonies and rituals. These interactions showcase the diverse human-nature relationships, fostering a sense of reverence and identity. By honouring these connections, communities preserve traditional knowledge, cultural heritage, and spiritual values, contributing to biodiversity conservation.



60. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 5% quantile

Please select an option ▾



61. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 95% quantile

Please select an option ▾



62. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 50% quantile

Please select an option ▾



63. Please explain your thinking behind these 3 answers



64. What effect will beavers have on the ability of woodlands to deliver these interactions? 5% quantile

Please select an option ▾



65. What effect will beavers have on the ability of woodlands to deliver these interactions? 95% quantile

Please select an option ▾



66. What effect will beavers have on the ability of woodlands to deliver these interactions? 50% quantile

Please select an option ▾



67. Please explain your thinking behind these 3 answers

Physical and experiential interactions

- Encompasses a wide range of experiences people have with ecosystems, like watching whales or birds, snorkelling, hiking, and leisure fishing or hunting. These interactions allow individuals to connect with nature, enjoy its beauty, and engage in recreational activities. By enabling these experiences, ecosystems promote physical and mental well-being, outdoor recreation, and tourism. They also nurture an appreciation for nature and support local economies, cultures, and biodiversity conservation efforts.



60. What effect will beavers have on the ability of inland service waters to deliver physical and experiential interactions? 5% quantile

Please select an option ▾



61. What effect will beavers have on the ability of inland service waters to deliver physical and experiential interactions? 95% quantile

Please select an option ▾



62. What effect will beavers have on the ability of inland service waters to deliver physical and experiential interactions? 50% quantile

Please select an option ▾



63. Please explain your thinking behind these 3 answers



64. What effect will beavers have on the ability of woodlands to deliver physical and experiential interactions? 5% quantile

Please select an option ▾



65. What effect will beavers have on the ability of woodlands to deliver physical and experiential interactions? 95% quantile

Please select an option ▾



66. What effect will beavers have on the ability of woodlands to deliver physical and experiential interactions? 50% quantile

Please select an option ▾



67. Please explain your thinking behind these 3 answers

⦿ **Heritage, cultural, scientific, entertainment, aesthetic, and educational interactions**

Covers a wide range of human experiences with ecosystems. This includes using ecosystems for research, learning, and entertainment, both in-person and through various media. Natural areas often hold important heritage sites and cultural artefacts, offering insights into history and culture. They also provide opportunities for hands-on learning about biodiversity and ecology. Additionally, ecosystems inspire art, connect us to nature, and offer spiritual experiences. By providing diverse opportunities for these interactions, ecosystems enhance human experiences, promote cultural diversity, and encourage conservation efforts.



76. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 5% quantile

Please select an option ▾



77. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 95% quantile

Please select an option ▾



78. What effect will beavers have on the ability of inland surface waters to deliver these interactions? 50% quantile

Please select an option ▾



79. Please explain your thinking behind these 3 answers



80. What effect will beavers have on the ability of woodlands to deliver these interactions? 5% quantile

Please select an option ▾



81. What effect will beavers have on the ability of woodlands to deliver these interactions? 95% quantile

Please select an option ▾



82. What effect will beavers have on the ability of woodlands to deliver these interactions? 50% quantile

Please select an option ▾



83. Please explain your thinking behind these 3 answers

⦿ **Existence and willingness to preserve and protect**

- ⦿ Valuing and protecting nature = Recognizing the inherent worth of wildlife, wilderness, and landscapes, as well as the joy and inspiration they bring. It involves wanting to safeguard these natural treasures for future generations, driven by moral and ethical concerns. This concept reflects a sense of responsibility toward the environment, emphasizing the desire to ensure its continued existence and enjoyment for all. In essence, it stresses the importance of preserving natural heritage for the benefit of humanity and the planet.



84. What effect will beavers have on the ability of inland surface waters to deliver existence and bequest services? 5% quantile

Please select an option ▾



85. What effect will beavers have on the ability of inland surface waters to deliver existence and bequest services? 95% quantile

Please select an option ▾



86. What effect will beavers have on the ability of inland surface waters to deliver existence and bequest services? 50% quantile

Please select an option ▾



87. Please explain your thinking behind these 3 answers



88. What effect will beavers have on the ability of woodlands to deliver existence and bequest services? 5% quantile

Please select an option ▾



89. What effect will beavers have on the ability of woodlands to deliver existence and bequest services? 95% quantile

Please select an option ▾



90. What effect will beavers have on the ability of woodlands to deliver existence and bequest services? 50% quantile

Please select an option ▾



91. Please explain your thinking behind these 3 answers

⦿ **Mediation of waste, toxins and other nuisances (by biota)**

- This category looks at how living organisms in ecosystems help manage waste, toxins, and other problems. It highlights how biota can interact with pollutants by breaking them down, storing them, or helping them degrade. Examples include microbes breaking down pollutants in soil, plants absorbing metals from the environment, and animals signalling environmental issues. For beavers this would include physiological, behavioural, and ecological interactions with pollutants and waste materials.



92. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by biota)? 5% quantile

Please select an option ▾



93. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by biota)? 95% quantile

Please select an option ▾



94. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by biota)? 50% quantile

Please select an option ▾



95. Please explain your thinking behind these 3 answers



96. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by biota)? 5% quantile

Please select an option ▾



97. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by biota)? 95% quantile

Please select an option ▾



98. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by biota)? 50% quantile

Please select an option ▾



99. Please explain your thinking behind these 3 answers

⊙ **Mediation of waste, toxins and other nuisances (by ecosystems)**

This category describes how ecosystems naturally manage and lessen the effects of waste, toxins, and other harmful substances. They do this through processes like filtering, absorbing, breaking down, and diluting pollutants. Ecosystems also help reduce smells and noise. Examples include wetlands filtering water contaminants, forests absorbing air pollutants and reducing noise, and coastal areas buffering against storms and pollutants.

100. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by ecosystem)? 5% quantile

Please select an option

101. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by ecosystem)? 95% quantile

Please select an option

102. What effect will beavers have on the ability of inland surface waters to mediate wastes, toxins and other nuisances (by ecosystem)? 50% quantile

Please select an option

104. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by ecosystem)? 5% quantile

Please select an option

105. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by ecosystem)? 95% quantile

Please select an option

106. What effect will beavers have on the ability of woodlands to mediate wastes, toxins and other nuisances (by ecosystem)? 50% quantile

Please select an option

107. Please explain your thinking behind these 3 answers

⦿ **Mediation of mass flows and erosion**

This category covers how ecosystems protect against erosion, landslides, and gravity flows. Vegetation holds soil in place with roots, while wetlands and forests act as barriers against mass movements. Additionally, ecosystems contribute to sediment transport and storage, with rivers and coastal areas redistributing sediment and wetlands trapping it, thereby enhancing landscape stability and resilience



108. What effect will beavers have on the ability of inland surface waters to mediate mass flows and erosion? 5% quantile

Please select an option ▾



109. What effect will beavers have on the ability of inland surface waters to mediate mass flows and erosion? 95% quantile

Please select an option ▾



110. What effect will beavers have on the ability of inland surface waters to mediate mass flows and erosion? 50% quantile

Please select an option ▾



111. Please explain your thinking behind these 3 answers



112. What effect will beavers have on the ability of woodlands to mediate mass flows and erosion? 5% quantile

Please select an option ▾



113. What effect will beavers have on the ability of woodlands to mediate mass flows and erosion? 95% quantile

Please select an option ▾



114. What effect will beavers have on the ability of woodlands to mediate mass flows and erosion? 50% quantile

Please select an option ▾



115. Please explain your thinking behind these 3 answers

⦿ Mediation of liquid flows

- ⦿ This category involves ecosystem services that regulate water flows and protect against floods. Ecosystems maintain baseline water levels for supply, recharge groundwater, and alleviate droughts and water shortages. They do this by capturing and storing rainwater through forests, wetlands, and permeable surfaces. Additionally, ecosystems absorb excess water, lessen peak flows, and provide natural flood buffers through vegetation and land cover.



116. What effect will beavers have on the ability of inland surface water to mediate liquid flows? 5% quantile

Please select an option ▾



117. What effect will beavers have on the ability of inland surface water to mediate liquid flows? 95% quantile

Please select an option ▾



118. What effect will beavers have on the ability of inland surface water to mediate liquid flows? 50% quantile

Please select an option ▾



119. Please explain your thinking behind these 3 answers



120. What effect will beavers have on the ability of woodlands to mediate liquid flows? 5% quantile

Please select an option ▾



121. What effect will beavers have on the ability of woodlands to mediate liquid flows? 95% quantile

Please select an option ▾



122. What effect will beavers have on the ability of woodlands to mediate liquid flows? 50% quantile

Please select an option ▾



123. Please explain your thinking behind these 3 answers

⦿ **Mediation of gas/air flows**

- ⦿ This category involves ecosystem services that regulate air movements, protect against storms, and aid ventilation and evapotranspiration. Vegetation acts as shelter belts, reducing wind speed during storms and lessening damage to landscapes and infrastructure. It also promotes air ventilation, reducing stagnation and pollution. Plants release water vapor through evapotranspiration, regulating humidity levels and local climate.



124. What effect will beavers have on the ability of inland surface water to mediate gas/air flows? 5% quantile

Please select an option ▾



125. What effect will beavers have on the ability of inland surface water to mediate gas/air flows? 95% quantile

Please select an option ▾



126. What effect will beavers have on the ability of inland surface water to mediate gas/air flows? 50% quantile

Please select an option ▾



127. Please explain your thinking behind these 3 answers



128. What effect will beavers have on the ability of woodland to mediate gas/air flows? 5% quantile

Please select an option ▾



129. What effect will beavers have on the ability of woodland to mediate gas/air flows? 95% quantile

Please select an option ▾



130. What effect will beavers have on the ability of woodland to mediate gas/air flows? 50% quantile

Please select an option ▾



131. Please explain your thinking behind these 3 answers

⊙ **Soil formation and composition**

Ecosystems provide services in forming and maintaining soil fertility and composition. Ecosystems contribute through processes like weathering, decomposition, and nutrient cycling. They maintain soil conditions by storing nutrients, decomposing organic matter, and fixing nitrogen. Healthy soils support plant growth, biodiversity, and agricultural productivity. They also regulate water availability, sequester carbon, and cycle nutrients, enhancing ecosystem resilience and sustainability.



132. What effect will beavers have on soil formation and composition in inland surface waters? 5% quantile

Please select an option ▾



133. What effect will beavers have on soil formation and composition in inland surface waters? 95% quantile

Please select an option ▾



134. What effect will beavers have on soil formation and composition in inland surface waters? 50% quantile

Please select an option ▾



135. Please explain your thinking behind these 3 answers



136. What effect will beavers have on soil formation and composition in woodland? 5% quantile

Please select an option ▾



137. What effect will beavers have on soil formation and composition in woodland? 95% quantile

Please select an option ▾



138. What effect will beavers have on soil formation and composition in woodland? 50% quantile

Please select an option ▾



139. Please explain your thinking behind these 3 answers

⦿ **Global, regional and micro climate regulation**

Ecosystems provide crucial services in controlling climate conditions globally, regionally, and locally. They help capture greenhouse gases like carbon dioxide, moderating global climate. Ecosystems also affect temperature, humidity, and wind, improving rural and urban climates. By regulating these factors, ecosystems influence regional weather patterns, shaping local conditions and climate resilience. Through processes like carbon capture, evapotranspiration, and cloud formation, ecosystems help stabilize climate and mitigate climate change effects.



140. What effect will beavers have on the ability of inland surface waters to regulate global, regional and micro climates? 5% quantile

Please select an option ▾



141. What effect will beavers have on the ability of inland surface waters to regulate global, regional and micro climates? 95% quantile

Please select an option ▾



142. What effect will beavers have on the ability of inland surface waters to regulate global, regional and micro climates? 50% quantile

Please select an option ▾



143. Please explain your thinking behind these 3 answers



144. What effect will beavers have on the ability of woodlands to regulate global, regional and micro climates? 5% quantile

Please select an option ▾



145. What effect will beavers have on the ability of woodlands to regulate global, regional and micro climates? 95% quantile

Please select an option ▾



146. What effect will beavers have on the ability of woodlands to regulate global, regional and micro climates? 50% quantile

Please select an option ▾



147. Please explain your thinking behind these 3 answers

⑥ Pollination and seed dispersal

- These ecosystem services involve different organisms like insects, birds, and mammals. Pollination, mainly by insects, is vital for plants to reproduce and produce fruits and seeds, supporting biodiversity and agriculture. Seed dispersal, done by insects, birds, and animals, helps plants spread across landscapes, boosting ecosystem resilience and vegetation growth in various habitats.



148. What effect will beavers have on the pollination and seed dispersal services of inland surface waters? 5% quantile

Please select an option ▾



149. What effect will beavers have on the pollination and seed dispersal services of inland surface waters? 95% quantile

Please select an option ▾



150. What effect will beavers have on the pollination and seed dispersal services of inland surface waters? 50% quantile

Please select an option ▾



151. Please explain your thinking behind these 3 answers



152. What effect will beavers have on the pollination and seed dispersal services of woodlands? 5% quantile

Please select an option ▾



153. What effect will beavers have on the pollination and seed dispersal services of woodlands? 95% quantile

Please select an option ▾



154. What effect will beavers have on the pollination and seed dispersal services of woodlands? 50% quantile

Please select an option ▾



155. Please explain your thinking behind these 3 answers

⦿ **Maintenance of nursery populations and habitats**

Ecosystems provide safe places for plants and animals to breed and raise their young. These areas, like seagrasses in the ocean and microstructures in rivers, are vital for young organisms. They offer protection, food, and ideal conditions for growth, ensuring the survival of different species and enhancing ecosystem resilience and stability.



156. What effect will beavers have on the maintenance of nursery populations and habitats in inland surface waters? 5% quantile

Please select an option ▾



157. What effect will beavers have on the maintenance of nursery populations and habitats in inland surface waters? 95% quantile

Please select an option ▾



158. What effect will beavers have on the maintenance of nursery populations and habitats in inland surface waters? 50% quantile

Please select an option ▾



159. Please explain your thinking behind these 3 answers



160. What effect will beavers have on the maintenance of nursery populations and habitats in woodlands? 5% quantile

Please select an option ▾



161. What effect will beavers have on the maintenance of nursery populations and habitats in woodlands? 95% quantile

Please select an option ▾



162. What effect will beavers have on the maintenance of nursery populations and habitats in woodlands? 50% quantile

Please select an option ▾



163. Please explain your thinking behind these 3 answers

🔍 Pest and disease control

- Ecosystems help manage pests, diseases, and invasive non-native species. This service is crucial for keeping landscapes healthy and productive. Ecosystems regulate pest and disease populations using natural predators, pathogens, and competition among species.
- They also prevent the spread of invasive non-native species, maintaining balance in the environment. By doing so, ecosystems protect ecosystems, support agriculture, and safeguard human health.



164. What effect will beavers have on the ability of inland surface waters to control pests and disease? 5% quantile

Please select an option ▾



165. What effect will beavers have on the ability of inland surface waters to control pests and disease? 95% quantile

Please select an option ▾



166. What effect will beavers have on the ability of inland surface waters to control pests and disease? 50% quantile

Please select an option ▾



167. Please explain your thinking behind these 3 answers



168. What effect will beavers have on the ability of woodlands to control pests and disease? 5% quantile

Please select an option ▾



169. What effect will beavers have on the ability of woodlands to control pests and disease? 95% quantile

Please select an option ▾



170. What effect will beavers have on the ability of woodlands to control pests and disease? 50% quantile

Please select an option ▾



171. Please explain your thinking behind these 3 answers

⦿ **Maintenance of water's chemical condition**

Ecosystems regulate and balance the chemical makeup of freshwater bodies, including the water itself and the sediment. Ecosystems ensure suitable conditions for aquatic life by processes like denitrification and nutrient cycling, which remove excess nutrients and pollutants, preventing harmful effects like eutrophication and algal blooms. They also stabilize pH levels and maintain chemical balance, supporting diverse aquatic life and functions. By regulating water chemistry, ecosystems sustain biodiversity, provide clean water for human use, and support ecosystem services.



172. What effect will beavers have on the ability of inland surface waters to maintain the chemical condition of water? 5% quantile

Please select an option ▾



173. What effect will beavers have on the ability of inland surface waters to maintain the chemical condition of water? 95% quantile

Please select an option ▾



174. What effect will beavers have on the ability of inland surface waters to maintain the chemical condition of water? 50% quantile

Please select an option ▾



175. Please explain your thinking behind these 3 answers



176. What effect will beavers have on the ability of woodlands to maintain the chemical condition of water? 5% quantile

Please select an option ▾



177. What effect will beavers have on the ability of woodlands to maintain the chemical condition of water? 95% quantile

Please select an option ▾



178. What effect will beavers have on the ability of woodlands to maintain the chemical condition of water? 50% quantile

Please select an option ▾



179. Please explain your thinking behind these 3 answers

Appendix D

	Calibration Score	Information score		
		All	Seed	Combined score
Regulating services				
EQ	0.6552	0.2492	0.3942	0.2583
GL 0.1	0.423	0.3072	0.5646	0.2389
GL 0.05	0.423	0.3041	0.5618	0.2377
GL 0.01	0.423	0.2992	0.543	0.2297
GL Opt	0.6789	0.9722	1.031	0.6996
Regulating + Cultural services				
EQ	0.2895	0.2313	0.3101	0.08979
GL	0.1135	0.3214	0.5475	0.06213

Appendix E

Ecosystem service	5%	50%	95%
Physical and experiential interactions	-0.41	0.28	0.78
Symbolic, sacred and/or religious interactions	-0.38	0.25	0.79
Heritage, scientific and educational interactions	-0.06	0.45	0.81
Aesthetic and entertainment interactions	-0.06	0.45	0.81
Existence and willingness to preserve and protect	-0.1	0.55	0.93
Mediation of waste, toxins and other nuisances (by ecosystems)	-0.03	0.24	0.64
Mediation of mass flows and erosion	-0.84	0.2	0.88
Mediation of liquid flows	0.01	0.28	0.87
Mediation of gas/air flows	-0.19	0.09	0.66
Soil formation and composition	0	0.39	0.82
Global, regional and micro climate regulation	-0.17	0.14	0.83
Pollination and seed dispersal	-0.09	0.17	0.85
Maintenance of nursery populations and habitats	-0.1	0.4	0.96
Pest and disease control	-0.4	0.11	0.65
Maintenance of water's chemical condition	-0.14	0.29	0.89

Appendix F

Variable	Min	Max	Mean	Median	SD
Age (years)	4.00	165.00	59.98	40.00	56.28
Area (m ²)	122.00	16277.00	3602.18	1188.00	4950.21
Average sediment depth (m)	0.71	56.27	17.99	12.47	15.16
Average water depth (m)	28.60	287.37	99.40	69.36	68.56
Conductivity (µS/cm)	40.00	419.00	156.27	118.45	102.67
Dissolved Oxygen (mg/L)	0.29	15.28	8.53	9.10	2.98
24hr temperature average (°C)	7.80	16.40	11.71	11.90	2.62
Total dissolved carbon (mg/L)	7.12	56.86	22.00	20.85	8.35
Total dissolved nitrogen (mg/L)	0.16	4.40	0.88	0.65	0.77
Total dissolved phosphorus (mg/L)	17.68	737.60	130.77	74.06	138.67
pH	6.65	8.36	7.60	7.60	0.39
24hr air pressure average (hPa)	999.00	1049.00	1017.37	1019.00	12.58

Variable	May	June	July	August	September
Conductivity (µS/cm)	165.54 ± 98.31	153.34 ± 95.9	160.61 ± 103.32	159.2 ± 134.69	143.94 ± 103.93
Dissolved Oxygen (mg/L)	9.91 ± 1.85	8.82 ± 2.13	7.34 ± 3.23	6.76 ± 3.69	9.03 ± 3.33
24hr temperature average (°C)	9.77 ± 1.39	9.97 ± 2.64	12.18 ± 1	15.84 ± 1.05	12.64 ± 1.96
Total dissolved carbon (mg/L)	17.56 ± 4.16	24.15 ± 6.29	22.21 ± 8.23	23.91 ± 14.29	22.99 ± 8.41
Total dissolved nitrogen (mg/L)	0.9 ± 0.86	0.81 ± 0.79	0.71 ± 0.28	0.65 ± 0.3	1.24 ± 1.05
Total dissolved phosphorus	72.65 ± 51.97	95.16 ± 82.79	164.98 ± 145.48	217.82 ± 191.86	141.94 ± 175.11
pH	7.71 ± 0.33	7.65 ± 0.33	7.67 ± 0.45	7.26 ± 0.42	7.54 ± 0.34
24hr air pressure average (hPa)	1014.5 ± 10.91	1017.72 ± 15.15	1012.39 ± 10.24	1016.1 ± 10.96	1025.56 ± 11.39

Appendix G

Response	Predictor	Estimate	Std.Error	Crit.Value	P.Value	Std.Estimate
Dissolved CO ₂	Total Dissolved Carbon	0.0921	0.0498	1.8473	0.0691	0.259
Dissolved CO ₂	Total Dissolved Nitrogen	-0.0285	0.0365	-0.7816	0.4372	-0.0802
Dissolved CO ₂	Total Dissolved Phosphorus	0.0317	0.0354	0.8952	0.3738	0.087
Dissolved CO ₂	pH	-0.118	0.0329	-3.5898	0.0006	-0.3316
Dissolved CO ₂	Conductivity	-0.0024	0.0496	-0.0479	0.962	-0.0067
Dissolved CO ₂	24hr Air temperature average	-0.0342	0.0321	-1.0671	0.2897	-0.0948
Dissolved CO ₂	24hr Air pressure average	0.1069	0.0322	3.3195	0.0015	0.3005
Dissolved CO ₂	Area of the pond	-0.0385	0.049	-0.7867	0.4342	-0.1083
Dissolved CO ₂	Dissolved oxygen	-0.0607	0.042	-1.4472	0.1524	-0.171
Dissolved CO ₂	Averaged sediment depth	0.1196	0.056	2.1377	0.0361	0.3365
Dissolved CO ₂	Averaged water depth	-0.1409	0.0523	-2.6926	0.0089	-0.3965
Dissolved CO ₂	Age	0.012	0.0562	0.213	0.832	0.0335
Flux CO ₂	24hr Air temperature average	-0.0805	0.056	-1.4365	0.1551	-0.0905
Flux CO ₂	24hr Air pressure average	-0.0761	0.063	-1.208	0.2309	-0.0869
Flux CO ₂	Area of the pond	-0.2272	0.0604	-3.764	0.0003	-0.2594
Flux CO ₂	Dissolved oxygen	-0.0215	0.0651	-0.3308	0.7418	-0.0246
Flux CO ₂	Total Dissolved Phosphorus	-0.1494	0.0582	-2.5662	0.0123	-0.1665
Flux CO ₂	Conductivity	-0.1184	0.0576	-2.0541	0.0435	-0.1351
Flux CO ₂	Dissolved CO ₂	1.9654	0.1923	10.2201	0.000	0.7984
Dissolved CH ₄	Total Dissolved Carbon	-0.0369	0.0957	-0.3857	0.7009	-0.0604
Dissolved CH ₄	Total Dissolved Nitrogen	-0.1723	0.07	-2.4615	0.0164	-0.282
Dissolved CH ₄	Total Dissolved Phosphorus	0.0047	0.0681	0.0693	0.9449	0.0075
Dissolved CH ₄	pH	-0.1291	0.0631	-2.0453	0.0447	-0.211
Dissolved CH ₄	Conductivity	0.0294	0.0953	0.3088	0.7584	0.0481
Dissolved CH ₄	24hr Air temperature average	0.0227	0.0616	0.3684	0.7137	0.0365
Dissolved CH ₄	24hr Air pressure average	0.0483	0.0618	0.7812	0.4374	0.079
Dissolved CH ₄	Area of the pond	0.0556	0.094	0.5918	0.5559	0.091
Dissolved CH ₄	Dissolved oxygen	-0.2716	0.0806	-3.3712	0.0012	-0.4448
Dissolved CH ₄	Averaged sediment depth	0.1569	0.1074	1.4605	0.1488	0.2568
Dissolved CH ₄	Averaged water depth	-0.189	0.1004	-1.8818	0.0641	-0.3095

Dissolved CH ₄	Age	0.0142	0.108	0.1312	0.896	0.0231
Flux CH ₄	24hr Air temperature average	-0.0045	0.0901	-0.0502	0.9601	-0.0044
Flux CH ₄	24hr Air pressure average	0.1617	0.09	1.7961	0.0765	0.161
Flux CH ₄	Area of the pond	-0.0436	0.0941	-0.4634	0.6444	-0.0434
Flux CH ₄	Dissolved oxygen	-0.0257	0.1056	-0.2438	0.8081	-0.0257
Flux CH ₄	Dissolved CH ₄	1.0208	0.1704	5.9897	0.000	0.6212

Enhancing Farmer Participation in Agri-Environmental Schemes in Scotland

Making the new support system work for farmers, nature, and climate

Scotland's future agricultural policy is undergoing significant reform through the Agriculture and Rural Communities (Scotland) Act 2024, which aims to position Scotland as a global leader in sustainable and regenerative agriculture. A cornerstone of this reform is a new four-tier agricultural support framework designed to deliver high-quality food production while addressing climate and nature challenges.

Agri-environmental schemes (AES) are central to this ambition. These programmes incentivise farmers and land managers to adopt practices that deliver public goods such as biodiversity enhancement, clean water, climate mitigation, and cultural landscape preservation. In Scotland, the main example is the Agri-Environment Climate Scheme (AECS). Under the new framework, Tier 1 provides foundational payments linked to Whole Farm Plans and basic environmental standards; Tier 2 will encourage more ambitious practices, such as reduced tillage and habitat conservation; Tier 3, building on AECS, will fund targeted environmental projects under longer-term agreements; and Tier 4 will deliver training and advisory support.

To meet Scotland's ambitious environmental goals, participation in AES must increase significantly. Evidence suggests that current schemes face key challenges around uptake and implementation. This briefing presents findings from a national survey of Scottish farmers and advisers, offering policy-relevant insights into the barriers and motivations shaping AES participation. These findings directly support the Scottish Government's aim of ensuring that the agricultural transition is inclusive, effective, and evidence based.

Key findings

- **High perceived risk:** Farmers viewed AES as financially uncertain, with concerns over whether applications would be successful, potential penalties for non-compliance, and the burden of long-term contracts.
- **Unclear application process:** Guidance and communication were widely regarded as confusing and inconsistent, leaving many farmers uncertain about requirements and feedback.
- **Inflexible scheme design:** Respondents highlighted narrow application windows, rigid contract terms, and limited autonomy over how land could be managed within AES agreements.
- **Strong environmental values, limited local knowledge:** While most participants cared deeply about environmental stewardship, many lacked confidence in identifying which wildlife improvements were most suitable for their local farming landscape.
- **Social support as a driver:** Family encouragement and community backing made farmers more likely to participate in AES and adopt wildlife improvements.
- **Education shapes engagement:** Respondents with higher levels of formal education reported greater environmental awareness and were more likely to engage with AES.

Study

This briefing draws on a co-designed national survey conducted in 2024, with 80 respondents including farmers, crofters, and agricultural advisers across Scotland. The survey was designed with farmer and adviser input to ensure clarity and relevance, and was distributed both online and in person at agricultural shows, markets, and through farming networks to reach a broad cross-section of the sector. It collected demographic information as well as attitudes towards AES, focusing on factors such as risk perception, flexibility, information access, and social support. Attitudes were measured using Likert scales - a common survey tool where respondents indicate their level of agreement or disagreement with a series of statements on a 0–5 scale. This allowed nuanced patterns in farmer perceptions and barriers to be analysed in a consistent, quantifiable way.

Recommendations / Implications for policy and practice

1. Empower farmer-led clusters via Tier 4 support

- Expand Tier 4 to fund voluntary, farmer-led clusters as part of training and advice provision.
- Clusters would deliver localised peer support, trusted advisory input, and landscape-specific training.
- Survey evidence shows social backing increases AES uptake; clusters would improve social acceptability, enhance environmental awareness, and builds trust.

2. Reduce risk and increase flexibility through Tier 2

- Make Tier 2 payments universally accessible and outcome-based, giving farmers flexibility in how they meet environmental objectives.
- Publish the rationale for tiered funding allocations to demonstrate transparency and ensure sufficient resources for AES.
- Offer optional contract lengths and allow regional adaptation of scheme design to reduce risk and improve accessibility, especially for risk-averse or tenant farmers.

3. Simplify and clarify the application process

- Introduce a streamlined online portal with clear guidance, built-in feedback, and responsive case officers, modelled on England's ELM scheme.
- Improve communication and feedback mechanisms to reduce administrative burden and frustration.
- Target improvements towards new entrants and advisers, who reported the lowest satisfaction with current application clarity.

About this research

This briefing is based on research undertaken by:

- David Bryan, PhD Researcher in Biological and Environmental Sciences, University of Stirling
- Dr Alan Law, Lecturer in Nature-Based Solutions, Faculty of Natural Sciences, University of Stirling
- Professor David M. Oliver, Professor of Catchment Science, Faculty of Natural Sciences, University of Stirling

For further detail, refer to the full research paper: *Enhancing participation in agri-environmental schemes (AES): A Scottish case study*. Available at University of Stirling DataSTORRE (<http://hdl.handle.net/11667/260>).

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SCOTFWAG
Scottish Farming and Wildlife Advisers' Group



Scottish
Government
gov.scot

Hydro Nation Scholars Programme

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August 2025

Enhancing Farmer Participation in Agri-Environmental Schemes in Scotland

Making the new support system work for farmers, nature, and climate



Deliver Sustainable Food Production



Support Climate And Nature Goals



Encourage Wider Participation In Agri-Environmental Schemes (AES)



Key Findings

High Risk:

Farmers see AES as financially uncertain, with fear of penalties and rigid contracts.

Unclear Process:

Guidance is confusing; requirements and feedback lack clarity.

Inflexible Design:

Narrow windows, rigid terms, and limited autonomy on land use.

Values vs. Knowledge:

Strong environmental commitment, but limited local know-how.

Social Support Matters:

Family and community backing boosts participation.

Education Drives

Engagement: Higher education links to greater awareness and uptake.

Recommendations / Implications for policy and practice

1. Support Farmer Clusters (Tier 4)



- Fund voluntary farmer-led clusters to provide peer support, trusted advice, and local training. Builds trust, awareness, and AES uptake.

2. Flexibility & Fairness (Tier 2)



- Make payments universal and outcome-based. Publish funding rationale, allow flexible contracts, and adapt schemes regionally to reduce risk.

3. Simplify Applications



- Create a clear online portal with guidance, feedback, and responsive officers. Cut admin burden and improve clarity for new entrants and advisers.

Appendix J



Farmer survey - Wildlife improvements

Information and consent

You are invited to participate in a survey investigating how economic, social, legal and institutional factors influence the uptake of wildlife improvements in farming landscapes. You have been invited to participate because you are **over 18** and **farm/manage and/or advise/consult on agricultural land in Scotland**. This survey is part of research project being conducted by David Bryan, a PhD student at the University of Stirling. It should take approximately **10 minutes** to complete.

Background Environment schemes encourage wildlife improvements that uses nature to solve problems. It's about finding clever and sustainable ways to work with nature to make our lives better. For example, instead of building a concrete flood wall, we might plant trees and create wetlands to soak up extra rainwater, which helps prevent flooding naturally. It's about using the environment's natural strengths, like trees, rivers, and wildlife, to tackle issues like climate change, flooding, and even improving our well-being. However, it is recognised that there are economic, social, legal and institutional factors that act as barriers stopping farmers and land managers from putting wildlife improvement's on their land. The aim of this research is to understand what these barriers are so that decision and policy makers can make it easier for farmers.

Do I have to take part? No. Your participation in this survey is voluntary. You may refuse to take part in the research or exit the survey at any time without penalty by pressing the 'Exit' button / closing the browser. You are free to decline to answer any particular personal information question for any reason – we have included a 'prefer not to say' option

Are there any benefits in taking part? No, there is no direct benefit

Are there any potential risks in taking part? There are no foreseeable risks involved in participating in this survey

Legal basis for processing personal data As part of the project we will be recording personal data relating to you. This will be processed in accordance with the General Data Protection Regulations

(GDPR). Under GDPR the legal basis for processing your personal data will be public interest/the official authority of the University.

What happens to the data I provide? Your answers will be completely anonymous, and we will use all reasonable endeavours to keep them confidential. Your data will be stored in a password-protected file and may be used anonymously in academic publications. Your IP address will not be stored. No identifying information would be included in any publications or presentations based on these data and your responses to this survey will remain confidential. Your personal data will be kept for two years on Research Drive – a secure data centre on the Stirling campus and then will be securely destroyed. Only the research team will have access to the research data.

Will the research be published? The research may be published in an academic journal and will be included as a chapter in my PhD Thesis. You will not be identifiable in any report/publication. The University of Stirling is committed to making the outputs of research publicly accessible and supports this commitment through our online open access repository STORRE. Unless funder/publisher requirements prevent us this research will be publicly disseminated through our open access repository. The results may also be presented at conferences. Furthermore, the work may be condensed into a policy brief for Scottish government

Who has reviewed this research project? The ethical approaches of this project have been approved via The University of Stirling General University Ethics Panel.

Your rights You have the right to withdraw from this project at any time without giving reasons and without consequences to you. You also have the right to object to us processing relevant personal data however, please note that once the data are being analysed and/or results published it may not be possible to remove your data from the study.

Whom do I contact if I have concerns about this study or I wish to complain? If you would like to discuss the research with someone then please contact David Bryan: dab11@stir.ac.uk You can also discuss the project with Dr David Oliver who is the lead PhD supervisor for this project: david.oliver@stir.ac.uk In case of complaint, you can contact the Head of the Biological and Environmental Science Department, Prof Kirsty Park: k.j.park@stir.ac.uk You have the right to lodge a complaint against the University regarding data protection issues with the Information Commissioner's Office (<https://ico.org.uk/concerns/>). The University's Data Protection Officer is Joanna Morrow, Deputy Secretary. If you have any questions relating to data protection these can be addressed to data.protection@stir.ac.uk in the first instance. Thank you for your time and for your participation in this project.

Do you consent to taking part in this survey? *

☐ Yes

☐ No

2. Do you currently farm or manage agricultural land in Scotland? *

☐ Yes

☐ No

Do you advise or consult for farmers and agricultural land? *

☐ Yes

☐ No

Getting to know you and your land

When we say environment schemes we are talking about agri-environment climate scheme (AECS), nature restoration fund, preparing for sustainable farming (PSF) and forestry grants etc. When we say wildlife improvements we mean measures that help with climate change and benefit nature.

What size is your farm? (Please specify hectares or acres)

What are the primary enterprises on your farm? (can select multiple)

- ☐ a. Arable e.g. cereals, oilseeds, potatoes and stockfeed
- ☐ b. Fruit e.g. berries, currents, orchard
- ☐ c. Vegetables e.g. peas, beans, leeks, turnips, carrots, lettuce etc.
- ☐ d. Dairy
- ☐ e. Beef
- ☐ f. Sheep
- ☐ g. Pigs
- ☐ h. Poultry
- ☐ i. Horticulture e.g. flowers, bulbs and nursery stocks
- ☐ j. Woodland
- ☐ k. Other (please specify)

If you select other, please specify

What is the Less Favoured Area (LFA) status of your farm?

- ☐ a. LFA
- ☐ b. Non-LFA

☐ c. Both

What is the ownership structure of your farm?

- ☐ Ownership
☐ Tenureship
☐ Both

How much land do you lease? (please specify hectares or acres)

How much land do you own? (please specify hectares or acres)

Is your holding a croft?

- ☐ Yes
☐ No

Which of these statements best describes your current position in relation to environment schemes? (e.g. agri-environment climate scheme (AECS), nature restoration fund, preparing for sustainable farming (PSF) and forestry grants).

- ☐ Currently not in a scheme and have no interest in joining
☐ Currently not in a scheme but interested in joining
☐ Currently in a scheme
☐ Currently not in a scheme but have been in the past
☐ Currently not in a scheme but have tried to join in the past

Are you a new entrant to farming?

- ☐ Yes

☐ No

What improvements for wildlife have you undertaken on your farm? (can select multiple)

- ☐ New/restored hedgerows
- ☐ rural sustainable drainage systems (e.g wetlands, ponds, sediment traps, bunds, swales)
- ☐ enhanced field margins (e.g. grass strips, water margins, conservation headlands,)
- ☐ arable managed for nature (e.g. wild bird seed mix, stubble retention, green manure, beetle banks)
- ☐ grassland managed for nature (e.g species rich grassland, grassland managed for birds)
- ☐ restoring natural rivers (e.g restoring river banks, floodplain management, remeandering, beaver reintroduction)
- ☐ management for habitat mosaic/scrub land
- ☐ woodland planting (e.g riparian, native woodland, agroforestry, silvopasture)
- ☐ Moorland management (e.g peatland restoration, sustainable grazing levels, grip/ditch blocking)
- ☐ Other

If you selected other, please specify

How important are the following reasons for your decision to farm?

Making a reasonable living

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Maximizing profitability

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Carrying on the family tradition

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Enjoying the farming way of life

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Looking after the countryside

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Being a progressive up to date farmer

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Keeping the land in 'good heart'

- ☐ Very important
- ☐ Important
- ☐ Neutral
- ☐ Somewhat unimportant
- ☐ Very unimportant

Where do you farm or advise?

- ☐ Argyll & the Islands
- ☐ Ayrshire
- ☐ Dumfries & Galloway
- ☐ East Central
- ☐ Forth & Clyde
- ☐ Highlands
- ☐ Lothians & Borders
- ☐ North East
- ☐ Orkney & Shetland
- ☐ Other

If you selected other, please specify

What is your main source of information about farm practices?

- ☐ Internet
- ☐ farm advisors
- ☐ family and friends

- ☐ farm magazines/publications
- ☐ Farm advisory service
- ☐ membership organisations (NFUS/PFLA/NFFN)
- ☐ other

If you selected other, please specify

When we say environment schemes we are talking about agri-environment climate scheme (AECS), nature restoration fund, preparing for sustainable farming (PSF) and forestry grants etc. When we say wildlife improvements we mean measures that help with climate change and benefit nature.

Awareness of environmental issue

I am well-informed of the environmental issues in farming

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

I know which wildlife improvements are most relevant for my local area

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

I believe adding more wildlife improvements is beneficial

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

I care for my local environment and feel a sense of stewardship

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Social acceptance

My community is supportive of environment schemes

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

There is no stigma attached to adopting wildlife improvements in my community.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

My family's approval motivates me to participate in environment schemes.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Positive influences from social media shape my opinion on environment schemes.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Risks

Adding wildlife improvements is a financial risk

- ☐ Strongly disagree

- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

It is risky applying for environment schemes because you might not be successful

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Environmental schemes are risky because of potential non-compliance

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

I worry about accountability if issues arise with wildlife improvements.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Environmental schemes have too much contractual risk

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Inflexibility

There is the risk of having to maintain wildlife improvements after the contract ends.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Contract lengths are not flexible enough

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Adding wildlife improvements feel like an irreversible decision

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Adding wildlife improvements limits my control over land use.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

The timeframe for applying to environment schemes is inconvenient.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree

- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Information

The cost of getting information (advisors, surveys, databases) on what wildlife improvements are suitable is

reasonable

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

There is enough support to help me make decisions about environment schemes

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Farmer meetings help me get the information I need.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

I get the information I need for environment schemes on time and it's adequate.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Applying for environment schemes is simple.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Environment scheme guidance is easy to understand

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Communication

The language in environment schemes is easy to understand.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Communication with environment schemes is positive and cooperative

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Communication with environment schemes is prompt and clear.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Case officers from environment schemes have a good understanding of farm requirements and operation

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree

- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

Feedback from environment schemes is clear and constructive

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neither agree nor disagree
- ☐ Agree
- ☐ Strongly agree
- ☐ Do not know/prefer not to say

What are the major difficulties that farmers face today and into the future?

Final questions

What is your highest level of education?

- ☐ High school
- ☐ College
- ☐ University

How would you describe your gender?

- ☐ Male
- ☐ Female
- ☐ Non binary/Third gender
- ☐ Prefer not to say

What is your age?

- ☐ 18-24
- ☐ 25-34
- ☐ 35-44
- ☐ 45-54
- ☐ 55-64
- ☐ 65+